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Future Forest Composition Under A Changing Climate And Adaptive Forest Management In Southeastern Vermont, Usa

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FUTURE FOREST COMPOSITION UNDER A CHANGING CLIMATE AND ADAPTIVE FOREST MANAGEMENT IN SOUTHEASTERN VERMONT, USA

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ABSTRACT

Global environmental change represents one of the greatest challenges facing forest resource managers today. The uncertainty and variability of potential future impacts related to shifting climatic and disturbance regimes on forest systems has led resource managers to seek out alternative management approaches to sustain the long-term delivery of forest ecosystem services. To this end, forest managers have begun incorporating adaptation strategies into resource planning and are increasingly utilizing the outcomes of forest landscape simulation and climate envelope models to guide decisions regarding potential strategies to employ. These tools can be used alongside traditional methods to assist managers in understanding the potential long-term effects of management and climate on future forest composition and productivity.

This study used a spatially explicit forest landscape simulation model, Landis-II, to examine and evaluate a range of long-term effects of current and alternative forest management under three projected climate scenarios within a 50,000-hectare forested landscape in southeastern Vermont, USA. Multiple scenarios were examined within this mixed ownership landscape, allowing for an evaluation of the influence of management and climate on future forest conditions in the region. These simulations indicate that land-use legacies and the inertia associated with long-term forest successional trajectories are projected to be an important driver of future forest composition and biomass conditions for the next 100 years. Nevertheless, climate is projected to have a greater influence on species composition and aboveground biomass over the next two centuries, with forests containing a greater abundance of species from more southerly regions and lower levels of aboveground biomass, resulting in shifts in the future provisioning of ecosystem services.

Key words*: Vermont, USA; climate change; forests; LANDIS-II; forest adaptation; forest management; above ground biomass; landscape inertia; land use recovery; forest composition*

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CHAPTER 1: ADAPTIVE FOREST MANAGEMENT

1.1 INTRODUCTION

Global environmental change is a term used to describe the composition of interacting historic and emerging agents of environmental change (Puettmann 2011) and represents one of the greatest challenges to natural resource management today. Anthropogenic climate change adds an additional level of complexity as managers look to manage the uncertainty around the future of forest ecosystems in the northeastern North America and around the world (Dale et al. 2001, Groffman et al. 2012). Given the mounting uncertainty around the future composition and function of our forests, resource managers and stakeholders are looking for tools and practical strategies to deal with these real and pressing challenges.

This chapter provides a review of the observed and projected impacts of global change on forest systems in the Northeast United States and synthesizes the management frameworks and tactics being researched and applied to address these challenges.

1.2 Managing for uncertainty

Understanding how forest systems respond to disturbance has been a major subject of ecological research and has led to advances in ecologically-based forest management (Holling 1973, Gunderson 2000). Disturbance shapes forest ecosystems by influencing their composition, structure, and function (Dale et al. 2001). Forest ecosystems and the species assemblages that define these systems have changed over time and are predicted to continue to change within the context of natural and novel disturbance regimes (Holling 1973, Gunderson 2000, Iverson et al. 2008, Duveneck et al. 2014). Insect pests, pathogens, and invasive plant species are among the primary agents of disturbance in North American forests (Dukes et al. 2009). In the northeastern United States these biotic agents of disturbance interact with small scale wind disturbances, land use change, shifting ownership regimes, and forest management decisions to form a complex system of forest disturbance. In the northeast, forest harvesting has been shown to be the primary disturbance agent within the landscape (Thompson et al. 2017) and is likely to continue to have significant role guiding future forest composition and functionality (Duveneck et al. 2017, Duveneck and Thompson 2019).

1.2.1 Forest disturbance and environmental change

Shifts in precipitation and temperature at local, regional, and global scales have been shown to influence the occurrence, timing, frequency, duration, extent, and intensity of disturbances (Dale et al. 2001, Turner 2010). Anthropogenic climate change is shifting global and regional climates beyond the natural range of variability observed over the last century (Hayhoe et al. 2007, Millar et al. 2007, Hayhoe et al. 2008, Puettmann 2011, IPCC 2013). In addition to climate change impacts, new anthropogenic stressors, including but not limited to, habitat fragmentation, land use changes, pollution, and the introduction of non-native plant and animal species and pathogens, are interacting with forest systems at varying scales (Millar et al. 2007). Non-native forest pathogens for example have been shown to have caused significant ecologic and economic damage over the past century and these impacts are projected to increase in the future (Lovett et al. 2016). Non-native forest pest, such as the hemlock woolly adelgid, whose range is currently limited due to cold winters, is projected to expand its range northward as climate warms leading to expected widespread mortality of eastern hemlock (Lovett et al. 2016)

Changing climate conditions are already altering disturbance regimes and suitable habitat for tree species (Dale et al. 2001, IPCC 2013). Annual temperature, precipitation, frequency of extreme events (flood, drought, storms), and growing season length are projected to increase in the Northeast over the next 50 years (Hayhoe et al. 2008). Soil moisture is projected to decrease due to increased evapotranspiration in the northeastern United States posing a greater risk of drought induced stress and potential negative impacts on regeneration of some desirable tree species (Hayhoe et al. 2008).

Under high emissions scenarios, tree species in the northeastern United States are projected to experience altered suitable habitat ranges (Iverson et al. 2008, Iverson and McKenzie 2013). In addition to direct impact to tree species, changing climatic conditions will likely increase indirect interactions with “nuisance” species such as insect pests, pathogens, and invasive plants (Dukes et al. 2009, Weed et al. 2013). The uncertainty around these potential forest compositional changes and their resultant effects on the provisioning of ecosystem services has led to growing consensus among resource managers that alternative and adaptive forest management approaches are needed (Millar et al. 2007, Spies et al. 2010, D'Amato et al. 2011, Puettmann 2011, Zhu et al. 2012, Janowiak et al. 2014).

1.2.2 Ecological resilience

Forest disturbances are dynamic and are increasingly being influenced by changing climatic conditions, shifting economic drivers, and social factors at local to global scales.

The challenges with sustaining the essential functions of forest ecosystems in a context of shifting and increasingly uncertain disturbance regimes has led forest managers and scientists to rely on concepts of ecological resiliency to assess the ability of a natural system to maintain critical ecosystem processes over time (Gunderson 2000, Cavers and Cottrell 2015).

The resilience of ecological systems was first described by C.S. Holling (1973) as "…the measure of the persistence of systems and of their ability to absorb change and disturbance and still maintain the same relationship between populations or state variables." *Ecological resilience* is more widely used and is defined as the "…amount of disturbance that a system can absorb without changing stability domains" or stable states (Gunderson 2000). Biological diversity and the diversity and redundancy of functional groups of species has been shown to play an important role in ecological resilience (Elmqvist et al. 2003). Another critical element to ecological resilience is the diversity of adaptive responses among functional groups of species to disturbance agents (Elmqvist et al. 2003). It is argued that the maintenance and promotion of ecological resilience and adaptive response diversity within forest systems should be a management objective of high priority when planning for future uncertainty (Gunderson 2000, Millar et al. 2007, Joyce et al. 2009, Puettmann 2011, Duveneck et al. 2014).

1.2.3 Ecological forestry

Ecological forestry arose from a demand for integrated forest management approaches that promoted ecological function and economic production while meeting other diverse goals and objectives (D'Amato et al. 2017b). One of the primary objectives of ecological forestry is to understand and work with natural patterns and processes to achieve management objectives (Seymour and Hunter, 1999). Through the use of silvicultural systems forest stands can be manipulated in ways that emulate the natural disturbance patterns of the region prior to extensive human alteration (Seymour and Hunter, 1999). Ecological forestry principles look to restore elements of natural forests by emulating the frequency, severity, and spatial pattern of disturbances with the hopes of conditioning these systems to respond favorably to human disturbances such as harvesting (Seymour 1999, Franklin 2007, Palik and D'Amato 2017).

The recognition of disturbance as a primary driver of ecosystem structure and function has led to the restructuring of silvicultural applications as natural disturbance emulating practices (O'Hara and Ramage 2013). These reframed applications seek to direct stands in ways that restore ecosystem functions and biological diversity (Seymour 1999, O'Hara and Ramage 2013), with retention of biological legacies often viewed as a strategy for increasing biodiversity conservation and carbon storage relative to traditional, production-oriented management scenarios (Donato et al. 2012, Gustafsson et al. 2012). Ecological forestry principles also promote a diversity of age structures and therefore allow for a greater diversity of tree species and habitat types to be present within a forest system or stand (Franklin 2007).

While ecological forestry has seen wide reaching application and has the ability to restore ecosystem function and foster increased resilience, a limiting factor in this approach is the dependence on the predictability of historical disturbance patterns. Today, natural disturbance regimes are being altered and are interacting with new disturbances with no historic analogue (Puettmann 2009). Many of the tenets and approaches of ecological forestry remain relevant to achieving forest resiliency, but may require reframing and modification to fully address the increased variability and uncertainty of future environmental change.

1.2.4 Adaptive silviculture

Increasingly, forest managers are tasked with managing for the uncertainty around the variability of future environmental change (Puettmann 2011). Historically, forest managers have relied on concepts of ecological sustainability, historical variability, and ecological integrity to determine management decisions (Millar et al. 2007). As new invasive pests and pathogens, shifting climatic conditions, and other novel environmental stressors interact with our forest systems, managing based solely on past forest conditions might limit long-term biodiversity conservation and ecological resiliency (Millar et al. 2007).

Adaptive silviculture has emerged to address these novel challenges. Building on the principles of ecological forestry, adaptive silviculture aims to sustain ecological function and economic productivity in the face of uncertain challenges. These approaches aim to use silvicultural treatments to promote resistance and resilience to change and, in some cases, aid in the transition of the system towards a state that may be better suited to projected future conditions (Millar et al. 2007). In order to maintain ecological integrity and economic productivity over time, adaptive silviculture looks to enhance the forest's adaptive capacity by focusing on managing the functional components of the system.

1.2.5 Adaptive silvicultural approaches

In order to incorporate the uncertainty around future forest conditions into resource planning, managers may need to rely more heavily on combining traditional and novel treatments and practices into new combinations to address current and future challenges (Millar et al. 2007). Silviculture has traditionally employed an iterative and adaptive process when evaluating the results of prescriptions. This historical integration of core adaptation principles will serve practitioners striving to achieve diverse management objectives under uncertain future conditions (Millar et al. 2007, Puettmann 2011, Janowiak et al. 2014). Millar et al. (2007) developed a framework of management which includes adaption and mitigation strategies that can be applied at the landscape and stand level. Options for adaptation under this framework (Figure 1) include measures which increase a forest system's ability to *resist* change, options which promotes the *resilience* of the system, and options which anticipate the expected change and assist in the *transition* the forest systems towards a state that is more adapted to projected future conditions (Millar et al. 2007).

Resistance measures are designed to protect the forest system from anticipated disturbances. These approaches are used where the forest system is of high economic, social, cultural, or ecological value and there is a desire or requirement that these systems be preserved as long as possible (Swanston et al. 2016). Tactics such as thinning to improve the growing conditions of desired species is an example of a resistance measure (D'Amato et al. 2013). These measures are often most effective in systems that have low vulnerability and are in a sense buffered from future changes. These approaches are best suited to meet short term objectives.

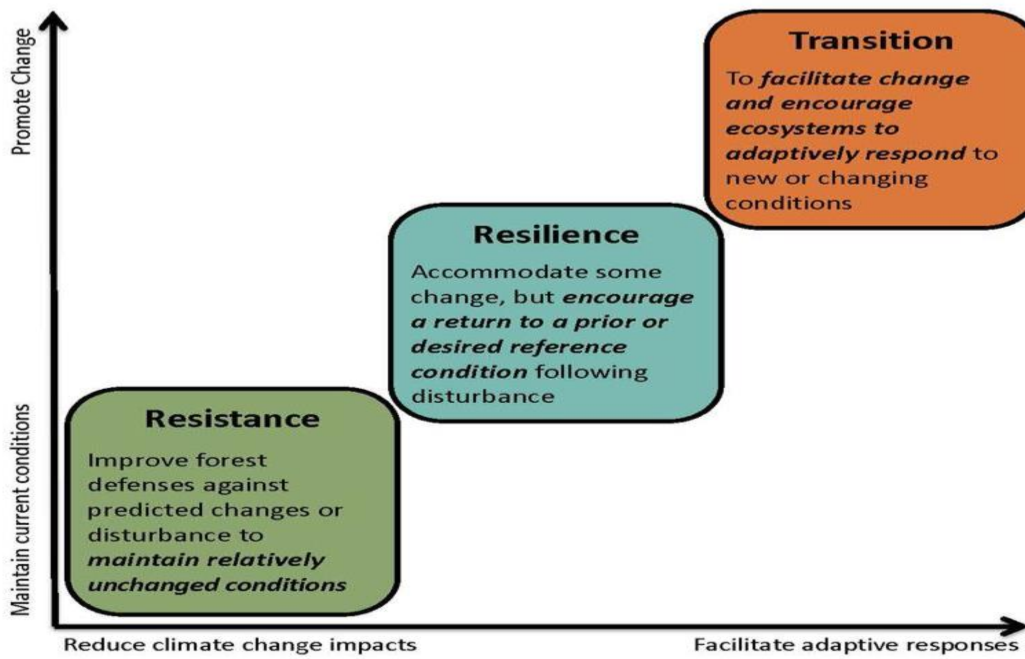

Figure 1: Conceptual framework for climate adaptation based on Nagel et al. (2018).

Management decisions which increase *resilience* anticipate potential impacts and promote the recovery of system function following a disturbance (Millar et al. 2007). Resilience measures enhance the system's ability to return to a desired state and maintain their function following a disturbance (Gunderson 2000). Tactics that use small scale disturbances as a means of increasing the diversity of species and age classes are examples of these measures (Spies et al. 2010, Janowiak et al. 2014, Nagel et al. 2017). These approaches are best suited for systems where there is moderate or high adaptive capacity present. The effectiveness of these approaches likely diminish, much like resistance measures, as the degree of change increases.

Transition approaches are designed to accommodate future change by assisting an adaptive response within the stand (Millar et al. 2007). While resistance and resilience actions focus on maintaining the current composition or function of the system, transitional actions anticipate these changes and look to enhance existing components of the system that are expected to do best under future environmental conditions. Transition tactics often aim to shift the composition of species, by natural or artificial means, to reflect changes in suitable habitat. These measures are often designed to meet long-term goals and are typically phased into management planning over time (Janowiak et al. 2014).

In addition to management actions that promote adaptation to projected environmental changes there are *mitigation* measures that can be employed. These measures aim to reduce greenhouse gas emissions by sequestering carbon on site in live and dead biomass (Millar et al. 2007, Malmsheimer et al. 2008, Evans and Perschel 2009, Ray et al. 2009, Nunery and Keeton 2010, D'Amato et al. 2011, Keeton et al. 2011).

The efficacy of these adaptive silvicultural treatments is currently being studied and evaluated across a wide range of forest conditions and geographical regions (D'Amato et al. 2011, 2013, Janowiak et al. 2014, Nagel et al. 2017, Ontl et al. 2018). In addition to long-term ecological studies and newly established experiments, scientists and managers are also looking to evaluate potential long-term impacts of management decisions under different climate scenarios. Increasingly, tools such growth and yield models, remote sensing tools, geographic information systems, and forest landscape simulation modeling have been used as additional tools to assist resource managers in long-term planning.

1.2.6 Decision support tools and forest landscape simulation models

Long-term and active silvicultural experiments offer great insights into the impacts and effectiveness of management decisions and can provide valuable perspectives for informing current management (D'Amato et al. 2011). While these experiments remain relevant as we look to re-tool management approaches for addressing global change, the uncertainty that remains around projected future climates has led many to look to computer technology to support traditional methods of resource management.

Forest managers are increasingly tasked with managing forest systems to meet diverse goals and objectives beyond the sustained production of wood products (Seymour, 1999). In doing so, forest managers are working across scales and using various tools to assess current forest conditions, project future changes, and develop systems to support complex management decisions. The recent advances in computer technology have led to further integration of spatial tools in the management and decision making process. These tools include, but are not limited to: mechanistic growth models, remote sensing (RS) and change detection (CD) techniques, geographic information systems (GIS), and forest landscape simulation models (FLSMs). Many of these tools can work in concert with each other leading to increasingly robust and useful applications.

Individual tree-based growth and yield models such as the Northeast–TWIGS and the Northeast variant of the Forest Vegetation Simulator (FVS–NE) have been used widely in the region (Nunery and Keeton 2010) and remain valuable resources. However, recent work suggests there are limitations to these models (Weiskittel et al. 2019). In addition, individual tree based growth and yield models remain limited in their ability to integrate stochastic events (i.e., disturbance) or the influence of climate variation on tree growth or establishment. However, stand or individual tree based mathematical models are increasingly being used to inform predictive and spatially explicit landscape scale models (Seidl et al. 2011).

As the scope of forest management planning continues to expand to larger scales, remote sensing techniques have proven to be a critical tool (Skidmore, 2011). Much of the research on ecological processes has been conducted at the plot or stand scale, as these approaches are often not economically feasible at larger scales. Recent advances in RS techniques have allowed resource managers to evaluate landscape dynamics across a wide range of spatial scales (Tewksbury, 2015).

Change detection uses remote sensing techniques to evaluate differences in the state of objects or phenomenon at different time steps (Hussain, 2013). By extracting remotely sensed images, change can be assessed quantitatively by evaluating the characteristics of objects or pixels at different time steps (Hussain, 2013). When applied at larger scales, these tools can be used to assess land use change or vegetation cover type change (Hansen et al. 2013). Furthermore, these tools can be integrated with GIS and used in the parameterization and validation of FLSMs.

GIS is one of the more powerful and accessible tools available to resource managers. GIS can be used to store, manipulate, analyze, and manage diverse sets of spatial data (Sani, 2015). As complex spatial data sets are now being integrated into information systems, these tools are now supporting decision making processes. Sani et al. (2015) outlined in detail how an integrated GIS can be used alongside RS to support multi-criteria decision making in the context of forest management. By using GIS, Sani et al. (2015) was able to assess a large forested land base and determine priority land uses for the landscape.

Finally, the use of forest landscape simulation models (FLSM) can be used at larger spatial scales as a tool to analyze ecological relationships and interactions (Seidl, 2011). FLSMs are computer programs used to project landscape change over time. Given the inherent complexity of ecological systems, FLSMs are useful in structuring quantitative analysis by bringing rich scientific knowledge to bear in management decision making (Seidl, 2011; Waring, 2007). Much of this work emerged from descriptive models that used empirical data to represent response variables and is now progressing to more processed-based approaches focusing on interaction between vegetation and disturbances across time a space (Seidl, 2011). All FLSMs are spatially explicit and often will use GIS to input, store, and display data (Scheller et al. 2007).

While the application of FLSMs continues to assist management decisions, uncertainties remain especially when simulating complex ecological systems in the context of a changing climate (Scheller 2018). Forest landscape simulation models should not be used as forecasts, however, they can provide quantitative insights into the range of future change under different climate scenarios and management regimes (Millar et al. 2007). The use of spatially explicit forest landscape simulation models such as LANDIS-II have been utilized in recent years as a means of providing fine-scale projections of forest compositional and functional changes under future climate conditions and forest management (Scheller et al. 2007, Ravenscroft et al. 2010, Duveneck et al. 2014, Duveneck and Scheller 2015, 2016). Outputs from these models in combination with feedback from scientists and resource managers have served as a central element in developing forest vulnerability assessments for much of the eastern US (Janowiak 2014).

The principles and frameworks highlighted in this chapter represent a range of the tools and tactics available to resource managers who seek to work with natural systems at varying scales while balancing multiple objectives in the face of unprecedented change and uncertainty.

CHAPTER 2: FUTURE FOREST COMPOSITION UNDER A CHANGING CLIMATE AND ADAPTIVE FOREST MANAGEMENT IN SOUTHEASTERN VERMONT, USA

2.1 INTRODUCTION

Global change represents one of the greatest challenges facing forest managers today (Millar et al. 2007). As managers continue to integrate multiple objectives into long-term planning, they face mounting uncertainty around future forest composition, productivity, and provisioning of goods and services due to shifting socioecological conditions (Puettmann 2011). Changing climatic conditions, altered disturbance regimes, increasing prevalence of invasive pests and pathogens, land use and tenure change, and shifting markets and societal demands on forest goods and services all add complexity to the management of forest ecosystems (Iverson et al. 2008, Zhu et al. 2012, Iverson et al. 2014, Iverson et al. 2017). This uncertainty and complexity has led resource managers, scientists, and policy makers to pursue further integration of vulnerability assessments (Janowiak et al. 2014, Janowiak 2018), adaptive management principles (Spies et al. 2010, D'Amato et al. 2011, Janowiak et al. 2014, Nagel et al. 2017), and decision support tools like forest landscape simulation models into forest management planning activities (Scheller et al. 2007, Ravenscroft et al. 2010, Duveneck and Scheller 2016).

Climate change is shifting global and regional temperature, precipitation, and disturbance regimes beyond the historic range of variability observed over the last two centuries (Millar et al. 2007, Puettmann 2011, IPCC 2013, Millar 2014, Janowiak 2018). Shifts in precipitation and temperature influence the occurrence, timing, frequency, duration, extent, and intensity of disturbances (Dale et al. 2001, Turner 2010), with resultant impacts on forest composition, structure, and function (Dale, Joyce et al. 2001). Climate trends over the last century in the northeastern United States show rising average annual temperature, increased precipitation, and a higher prevalence of extreme weather events (Rustad 2014, Janowiak 2018). These trends are projected to continue and intensify in the next century (Hayhoe et al. 2008, Janowiak 2018), resulting in significant changes to forest ecosystems, notably altered suitable habitat ranges for tree species (Iverson et al. 2008, Iverson and McKenzie 2013). It is predicted that forest types associated with higher elevation, colder climates (e.g., spruce-fir forest types) will see a decline in suitable habitat over the next century, while tree species adapted to lower elevation, warmer growing conditions, and southern latitudes (i.e. oak-hickory forest types) across the northeastern United States are projected to experience increased suitable habitat (Iverson et al. 2008, Iverson and McKenzie 2013, Iverson et al. 2017, Janowiak 2018). However, the degree to which tree species or forest community types will actually shift their range in response to these changes given the projected pace of shifting climatic conditions, disturbance regimes, and successional dynamics related to land use legacies remains uncertain (Zhu et al. 2012, Zhu et al. 2014, Foster and D'Amato 2015, Zolkos et al. 2015, Woodall et al. 2018).

While changing climate is projected to influence future forest composition and biomass conditions across the northeastern United States (Iverson et al. 2008, Thompson et al. 2011, Duveneck et al. 2017, Iverson et al. 2017, Wang et al. 2017, Ma et al. 2018), successional dynamics related to the recovery from historic, intensive land use (Foster et al. 1998), stand dynamics, and forest management are predicted to continue to influence future forest development (Thompson et al. 2011, D'Amato et al. 2013, Duveneck et al. 2017). For example, recent simulations of future forest dynamics for this region suggested that land-use legacies and resulting successional trajectories far outweighed the influence of climate on forest dynamics over the next century; however, it is unclear how these patterns might be influenced by adaptive management strategies designed to address climate impacts (Duveneck et al. 2017). The degree to which current and alternative forest management decisions can influence future forest conditions and climate adaptability and resiliency within a landscape of mixed ownerships remains a key knowledge gap.

In our study, we utilize a spatially explicit forest landscape simulation model (Landis-II version 6.2) to evaluate the impacts of climate change, forest disturbance, and a range of current and alternative forest management approaches on future forest composition within a landscape of mixed-ownership in southeastern Vermont, USA. In this study, we address the questions: (1) How are species composition and biomass conditions projected to change over a 200-year period within a landscape of private and public ownerships subject to changing climate and management regimes? and (2) How will levels of application of adaptive measures influence landscape-level resilience to climate change?

2.2 METHODS

2.2.1 Study area

The study was conducted within a 56,801-hectare forested landscape of mixed ownership in southeastern Vermont, USA (Figure 1). The study landscape is characterized by low-moderate elevation foothills (122-732 meters above sea level) within the watershed of the Connecticut River. The landscape is primarily forested with some areas of rich agricultural land in the valleys. The predominant natural forest community type found within this region is the “Northern Hardwood Forest” (Thompson 2000) with sugar maple (*Acer* *saccharum),* yellow birch (*Betula alleghaniensis),* American beech (*Fagus grandifolia),* eastern hemlock (*Tsuga canadensis),* and eastern white pine (*Pinus strobus)* being most common species within the landscape.

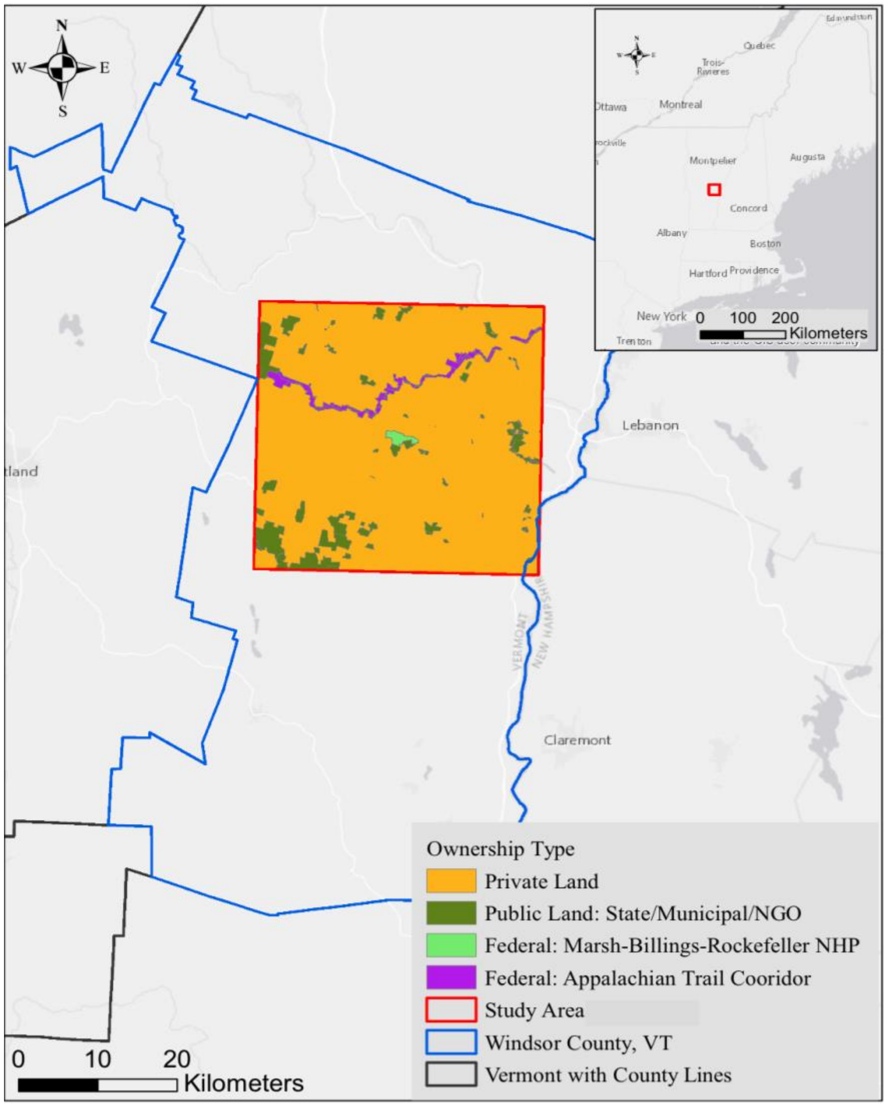

Figure 1: Study area located within Windsor County in southeastern Vermont. The simulated landscape (shown in red) is comprised of public and private ownerships and is primarily forested with small areas of agricultural and residential land.

Land ownership within the study area can be characterized as "mixed" with private, state, municipal, non-profit, and federal ownership classes present. Private land ownership comprises the majority of the study area (91%) and is characterized primarily by family forest owners (FFO). Public ownerships make up 5.8% of the landscape and are comprised of State, municipal, and non-profit ownerships (Table 1). The two main Federal ownerships in the study area are the Marsh-Billings-Rockefeller National Historical Park (MABI) and the Appalachian Trail (AT) corridor together, which comprise 3.1% of the study area. Given the distinct differences in management of these two ownerships, we have decided to treat each federal ownership as separate management areas in this study. MABI is one of the only National Historical Parks in the country that demonstrates active forest management for multiple benefits. The AT corridor is collaboratively managed with no active timber harvesting by the National Park Service, the Green Mountain Club, and private landowners. This mosaic of private, public, and federal ownerships provides a unique study area to analyze the long-term impacts of multiple management decisions within a single landscape.

Table 1: Ownership type as a proportion and percentage of the total forested area in the study area. *total forest area does not include non-forest land within study area. total land area = 56,800.80 hectares.

Ownership	**Area (ha)**	**Percent of forest land (%)**
Private	42,754.4	91.0%
Public	2,746.5	5.8%
Federal – MABI	187.9	0.4%
Federal – AT	1,290.7	2.7%
Total Forest Area	46,979.6*	100.0%

2.2.2 Simulation model and parametrization

For this study, we used LANDIS-II (v6.2), a spatially-explicit landscape simulation model to analyze the interactions between climate, timber harvesting, and natural disturbance (e.g. wind and forest tent caterpillar disturbance) within a predominately forested landscape. LANDIS-II simulates successional dynamics, seed dispersal, and response to disturbances such as harvesting (Scheller et al. 2007, Ravenscroft et al. 2010). Average aboveground biomass, annual net primary production (ANPP), deadwood biomass pools, and mortality are also simulated in this model. Successional dynamics and ecosystem processes are simulated within a landscape of interacting grid cells at a spatial resolution of 30x30 meters. All cells are grouped together into blocks or ecoregions based on similar topographic and edaphic properties. Tree species-age cohort information is initially spatially imputed from local continuous forest inventory plots located within the study area as has been done in previous studies (Ravenscroft et al. 2010, Duveneck et al. 2014, Duveneck and Scheller 2015).

We modeled 26 of the most abundant tree species within the landscape based on species importance valued derived from continuous forest inventory data from 1,530 plots within the Green Mountain National Forest located to the west of the study area and 144 plots located within MABI located at the center of the landscape. Species-specific attributes related to shade tolerance and seed dispersal distance were obtained from the United States Forest Service silvics manual for North American tree species (Burns 1990). Additional input parameters for max annual net primary production (MaxANPP) and probability of tree seedling establishment (Pest) for each species were calculated based on soil properties for each ecoregion using the LINKAGES and PnET-II generalized ecosystem process model (Aber et al. 1995). Maximum biomass (MaxB) for each species is calculated based on species specific relationships between MaxANPP and MaxB as has been done in previous studies (Thompson et al. 2011, Duveneck and Scheller 2016). Specific species input parameters are outlined in Table 2.

Table 2: Tree species life history attributes, probability of establishment as an average over all ecoregions for year 0 of the simulation, average MaxANPP across ecoregions for year 0 of the simulation, and average MaxBiomass across all ecoregions at year 0 simulation. Shade tolerance 1-5; 1=intolerant.

Species	Longevity (yrs)	Sexual Maturity (yrs)	Shade tolerance	Max seed dispersal (feet)	Vegetative Reproduction Probability	MaxANPP $(g \cdot m^{-2} \cdot yr^{-1})$	MaxBiomass $(g \cdot m^{-2})$
Abies balsamea	200	25	5	160	0.0	595	17,857
Acer pensylvanicum	100	15	5	100	0.1	645	6,453
Acer rubrum	150	10	4	200	0.5	1,076	32,272
Acer saccharum	350	40	5	200	0.1	1,391	41,739
Betula allegeniensis	300	40	3	400	0.1	1,281	38,424
Betula lenta	275	40	2	400	0.1	1,281	38,424
Betula papyrifera	100	30	1	2,000	0.5	1,367	41,005
Carya cordiformis	200	30	1	120	0.9	1,582	47,445
Carya ovata	300	40	2	120	0.8	1,582	47,445
Fagus grandifolia	350	40	5	150	0.9	1,164	34,922
Fraxinus americana	300	30	2	140	0.1	1,228	36,850
Larix decidua	180	15	1	200	0.0	197	29,910
Ostrya virginiana	150	25	4	1,000	0.4	1,132	33,957
Picia abies	200	30	5	100	0.0	288	14,400
Picia rubens	400	25	5	200	0.0	292	14,600
Pinus resinosa	300	35	2	275	0.0	488	14,640
Pinus strobus	400	40	3	210	0.0	1,040	31,214
Pinus sylvestris	200	10	2	2,000	0.0	642	19,260
Populus grandidentata	100	10	1	1,000	0.9	1,761	52,947
Populus tremuloides	100	20	1	1,000	0.9	1,458	43,737
Prunus pensylvanica	35	10	1	5,000	0.5	1,627	48,803
Prunus serotina	250	30	1	200	0.75	1,735	52,059
Quercus alba	400	50	3	1,500	0.5	1,426	42,793
Quercus rubra	250	25	3	1,500	0.5	1,183	35,481
Tilia americana	250	30	3	120	0.9	1,765	52,935
Tsuga canadensis	500	30	4	100	0.0	418	12,540

2.2.3 Experimental design

For this study, we simulated the landscape dynamics for 200-years at 5-year consecutive time steps. We utilized the following extensions to the LANDIS-II core framework: Biomass Succession (v5.02), Biomass Harvest (v4.0), Biomass Insects (v3.0), and Base Wind (v3.0) (Mladenoff 1999, Gustafson et al. 2000, Scheller and Mladenoff 2004, Foster 2011). We also utilized the extensions, Biomass Reclassification Output (v3.0) and Biomass Output (v3.0) for our analysis of compositional changes within the study landscape (Scheller and Mladenoff 2004). We simulated three forest management regimes under three climate regimes (current, low emissions, and high emissions) resulting in nine scenarios which were each replicated five times.

2.2.4 Climate data

Climate change was simulated within the model using NASA Earth Exchange (NEX) downscaled (800m resolution) climate projection models, which were derived from the General Circulation Model (GCM) runs developed for the Fifth Assessment Report for the Intergovernmental Panel on Climate Change (IPCC) (Moss et al. 2010). We utilize four GCMs (HADGEM2-AO, CESM1-BGC, CCSM4, and MPI-ESM-LR) and two Representative Concentration Pathways (RCP 4.5 and RCP 8.5) to represent the potential range of projected climate variability for the landscape for the years 2006-2099. Current climate was derived from historic PRISM climate data for the region from 1900-2017 (Daly et al. 2008). Daily average, minimum, and maximum future temperature, precipitation, and growing degree day projections were used, along with a suite of site-specific variables related to soil properties, as input parameters for the ecosystem process models PnET-II and LINKAGES. These models were used to calculate future probabilities of tree establishment and growth rates for all tree species in the study landscape. Because climate projections are only available for the next one-hundred years, the trend in probabilities of establishment and growth rates calculated for the last thirty years of this century were extrapolated into the next century of the simulation. This approach of extrapolation has been done in previous studies (Duveneck and Scheller 2016).

The ecosystem process models outlined above utilize fine scale climate data; however, in order to illustrate the variability in the climate models used in this study, we present the average annual July temperature and total annual precipitation for the four GCMs and each RCP for each year in the simulation for high (RCP-8.5) and low (RCP-4.5) emissions scenarios. Under high the emissions scenario mean July temperature is projected to increase by 6.8°C (Figure 2) and total annual precipitation is projected to increase by 19.1cm within the study area (Figure 2). Under the low emission scenario mean July temperature is projected to increase by 2.28°C (Figure 2) but total annual precipitation averaged across all ecoregions is not projected to change dramatically (Figure 2).

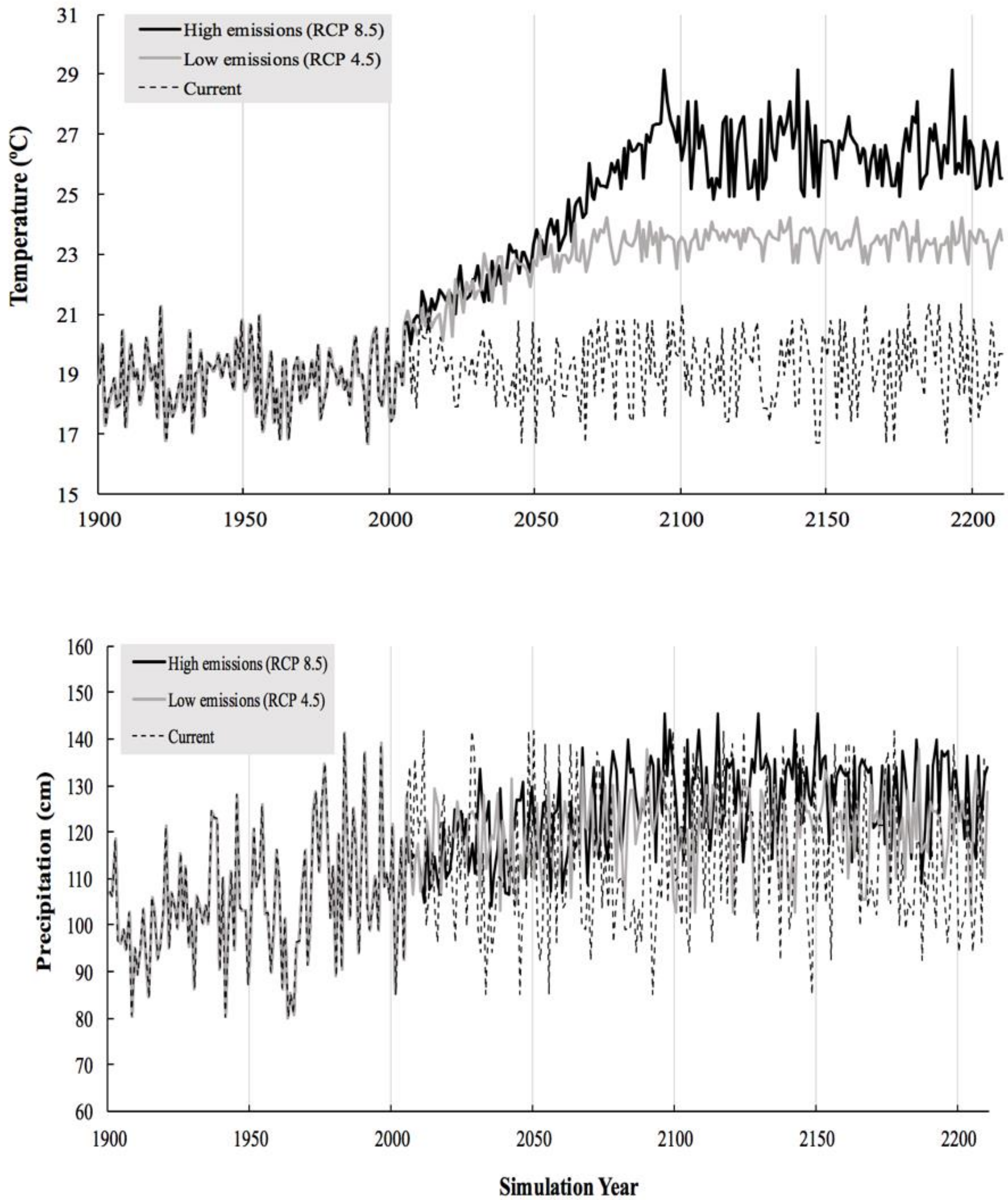

Figure 2: Mean July temperature (Top) and average annual precipitation (bottom) under current climate and two RCPs (4..5 and 8.5). Current climate for the period of 2017-2210 is based on a random sample of mean July temperature and average annual precipitation from 30 years prior to 2017

2.2.5 Management

Timber harvesting is one the most common disturbances and has the greatest impacts on mature tree mortality in the northeastern United States compared to other natural and anthropogenic disturbances (Canham et al. 2013). Harvesting influences forest composition, structure, and function and varies in intensity and silvicultural objectives across biophysical and social gradients (Kittredge et al. 2003, Kittredge et al. 2017, Thompson et al. 2017). In order to accurately represent current and future management decisions across ownerships for our study, we relied on publically available information (site specific management plans for public ownerships etc.) and expert opinion from consulting foresters and land managers working within the landscape. Input parameters for harvest frequency and intensity across all ownerships in the study area was determined based on previous studies relevant to the region (Kittredge et al. 2003, McDonald et al. 2006, Kittredge et al. 2017, Thompson et al. 2017) and an independent assessment of harvest volumes reported to the State of Vermont by wood product purchasing entities and facilities (e.g. sawmills, biomass facilities, etc.) from the past 10 years. These data are made publically available in annual harvest reports developed by the Vermont Agency of Natural Resources' Department of Forests, Parks and Recreation (https://fpr.vermont.gov/harvest-reports). In addition, we utilized two Landsat imagery data sets to assess forest disturbance patterns related to harvesting within the study area (Goward, 2016; Hansen, 2013).

Social and biophysical factors have been shown to influence harvesting regimes in the region (Kittredge et al. 2017) and Thompson et al. (2017) determined that ownership related factors were the most predictive of forest harvest intensity and frequency. Across all forest types and ownerships in the northeastern United States, it is estimated that approximately 2.6% of forest land is harvested every year (Thompson et al. 2017). This impact varies across ownership and region with private lands (2.9% $\cdot yr^{-1}$on private woodland owners and 3.6%$\cdot yr^{-1}$on private cooperate lands) predicted to be harvested more than State (1.6%$\cdot yr^{-1}$), Federal (1.0%$\cdot yr^{-1}$), and municipal lands (2.4%$\cdot yr^{-1}$) (Thompson et al. 2017).

Harvest Report Data: Windsor County, Vermont USA

Year	Saw/Veneer(Mg)	Pulp(Mg)	Fuel (Mg)	Total Harvest(Mg)
2016	48,032	29,351	167,926	245,309
2015	48,744	52,452	166,221	267,417
2014	48,009	42,634	143,191	233,834
2013	45,777	18,738	158,132	222,648
2011	94,435	23,871	72,673	190,979
2010	95,292	28,285	85,298	208,875
2009	125,225	29,266	64,105	218,597
2008	49,343	30,505	49,282	129,129
2007	130,405	20,793	44,001	195,200
2006	126,427	18,296	41,316	186,040
2005	151,053	21,934	39,492	212,480
Sum	962,743	316,124	1,031,637	2,310,504
Mean	87,522	28,739	93,785	**210,046**

Table 3: Annual harvest report data reported to the state of Vermont for harvested volumes in Windsor County, VT by wood product purchasing facilities (sawmills etc.) and used to determine a baseline harvest rate for the study area.

Through an assessment of annual harvest reports (Table 3) maintained by the State of Vermont Agency of Natural Resources' department of Forests, Parks, and Recreation, we determined that approximately an average of *39,360* metric tons (Mg) is harvested annually within our study area (Table 4). We calculated the average reported harvest volumes for the county in which our study area is located and calculated a proportional harvest volume based on the forest land area the study area compared to the forested area of the entire county.

Windsor County area (ha)	Windsor County mean annual harvest (Mg)	Study area forested land(ha)	Study area proportional annual mean harvest (Mg)
250,975	*210,046*	*46,871*	***39,227***

Table 4: Forested land in study area and harvest rate based on the proportion of the study area to the area of Windsor County, VT.

We utilized two previously developed spatial data sets to quantify disturbance patterns within our study area (Appendix III, IV). We used the North American Forest Dynamics/NASA Earth Exchange (NAFD-NEX) and the Global Forest Change (v.1.4) 2000-2016 spatial data sets (Goward, 2016; Hansen, 2013). The NAFD-NEX data set processes Landsat imagery to classify forest disturbance from year 1986 to year 2010 with a 30x30m resolution across the conterminous United States. The Global Forest Change data set classifies forest loss between the years of 2000 and 2016. These two data sets detect forest harvesting and, to a lesser degree, natural disturbance at a spatial resolution of 30x30m (0.09 ha) allowing the user to conduct a simple raster analysis in a GIS to assess disturbance patterns across a landscape. Disturbances such as partial harvesting, intermediate thinning treatments, and small-scale wind and insect events occur at spatial scales too small to detect and were therefore, not included in this assessment. However, these tools do provide insights into the spatial distribution of timber harvesting events that create larger openings in the canopy (e.g. group/patch selection treatments, shelterwood harvests, silvicultural clearcuts etc.).

Table 5: Patch size disturbance patterns for study area based on the Global Forest Change (Top) and NAFD – NEX data sets (Bottom)

Global Forest Change - Disturbance Area (2000-2016)				
Ownership	**n patches (0.09 ha)**	**Mean(Ha ± SE) Size**	**Median**	**Range**
Public	147	0.45 ±0.09	0.18	0.09 - 10.52
Private	2,715	0.30 ±0.01	0.18	0.09 - 27.53
Federal	11	0.16 ±0.03	0.09	0.09 - 0.45
Marsh-Billings	9	0.23 ±0.05	0.18	0.09 - 0.54
Total Landscape	2,882	0.31 ±0.01	0.18	0.09 - 27.54

NAFD-NEX Disturbance Area (1987-2010) -ACRES				
Ownership	**n patches (0.09 ha)**	**Mean(Ha ± SE) size**	**Median**	**Range**
Public	165	0.53 ±0.10	0.18	0.09 - 11.52
Private	4,842	0.47 ±0.02	0.18	0.09 - 18.36
Federal	86	0.52 ±0.18	0.18	0.09 - 14.94
Marsh-Billings	15	1.20 ±0.88	0.18	0.09 - 13.41
Total Landscape	5,108	0.48 ±0.02	0.18	0.09 - 32.40

Mean patch size (above detectable minimum of 0.09 ha) was largest within public lands (0.45 +/- 0.09 ha) compared to private (0.30 +/- 0.01 ha), Marsh-Billings (0.23 +/- 0.05), and federal lands (0.16 +/- 0.03 ha) based on the Global Forest Change data set (Table 8). The NAFD-NEX data set showed larger patch sizes across all ownerships (Table 5). These findings show disturbances larger than 0.09 ha in size occur on approximately

0.22% of the landscape annually and these disturbances are on average between 0.31ha and 0.48 ha in size. Given a predicted harvest disturbance rate for the region of approximately 2.6% annually (Thompson et al. 2017), our assessment suggests that partial harvesting and intermediate treatments may be underrepresented, as this is believed to be the predominate treatments in the region's current management regime (Keith Thompson, Sam Schneski, and Allen Follansbee, personal communication, October 25, 2017). Given private landownership is the predominate ownership type in our study area, we conducted an additional assessment of private landowner forest management practices (Appendix II). In addition to personal communication with land managers, we utilized publically available information from the State of Vermont's Use Value Appraisal (UVA) program. This program, analogous to "current use" programs in neighboring northeastern States, provides preferential tax assessment to private landowners who enroll forestland in the program. Landowners in this program make a commitment to sustainably manage their forest and restrict development and non-compatible uses for a 10-year period. Enrollment in the program can be renewed after each period, provided landowners comply with the program guidelines. All forestland that is enrolled in the program must have a forest management plan with specific silvicultural objectives outlined and planned management activities that are approved by the State of Vermont County Forester Program. Additionally, all forest harvesting activities on that forest land enrolled in UVA program must be reported to the State with detailed description of the forest type and area treated along with estimated volume of timber removed.

Through an assessment of current UVA forest management plan data, we determined that uneven-aged silvicultural approaches, primarily single tree selection, were prescribed most frequently (28%) followed by intermediate thinning treatments (12%) and group selection (8%) across all forest types enrolled in the program in Windsor County, Vermont (Figure 3).

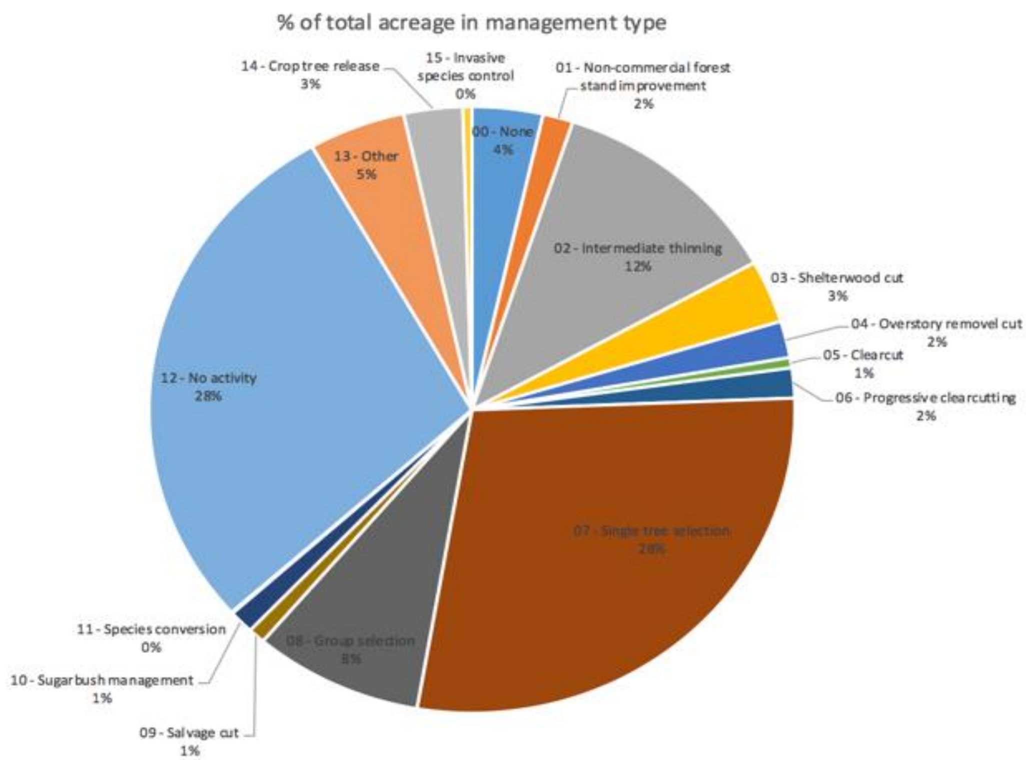

Figure 3: Primary silvicultural treatment type for UVA enrolled private forest land in Windsor County, Vermont.

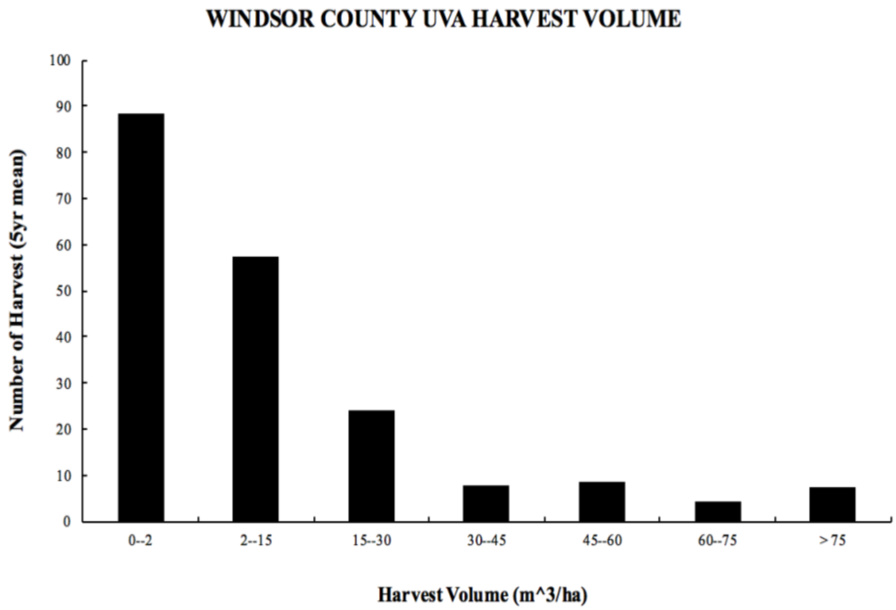

Figure 4: Distribution of harvest volumes for private landowners enrolled in the Vermont UVA program. Based on total area of parcel and not the area of treated area therefore provides a relative measure of harvest intensity. Values based on a five-year average of reported harvest volumes between the years of 2012-2016

Annual harvest rates were analyzed for Windsor County, Vermont using Forest Management Activities Report (FMAR) data. A subset of these data was used to determine the average timber volume removed per acre over a five-year period. This analysis was also used to inform parameterization of the harvest extension in the simulation model. The annual harvest rate from 2005-2016 in Windsor County was 8.8 $m^3 \cdot ha^{-1}$ (+/- 0.63) (Appendix IV, Figure 4;).

Because the harvest volumes reported here (Figure 4) are based on the total area of the property and not treatment area, they serve more as a relative measure of harvest intensity. These findings reflect our previous results that indicate that less intensive harvesting events are typical within privately owned land. The majority of harvest occurring are at a low intensity which would correspond to partial harvesting such as single tree selection, intermediate treatments thinning, or other partial harvesting.

Based on the findings of the assessments highlighted above and feedback from resource managers working within the region, we developed a set of silvicultural treatments (Table 6) with expected frequency and intensity for each ownership in the study area (Table 7). These treatments form the foundation of the current management scenario in the simulation. The biomass harvest extension in Landis-II allows users to design silvicultural treatments that can be carried out within the landscape according to a set of specific rules. Current management within the region is characterized by periodic partial harvesting such as single-tree selection occurring in stands dominated by hardwood species. Small group selection, intermediate thinning, and patch cutting also occur within the landscape under current management regimes.

Table 6: Silvicultural treatments simulated within the study area. *Cutting cycle is defined as the amount of time between subsequent harvest entries in an uneven-aged management prescription.

Silvicultural method	LANDIS-II Code	Description
Single tree selection	SingleTree1	Single tree selection opening on 20-year cutting cycle*. 20% of overstory removed at each entry. Stands with high component of sugar maple and ash will be treated preferentially.
Partial harvest	Partial	Randomly located harvest removing 30-40% of overstory. No cutting cycle is adhered to.
Group selection	GroupSelection1	0.1-0.2 hectare gaps created on 20-year cutting cycle. Located on sites where beech is more prevalent within Northern Hardwood stands.
Group selection	GroupSelection2	0.2 - 0.3 hectare gaps created on 20-year cutting cycle. Located on sites where beech is more prevalent. Located within northern hardwood and or on sites with a mixture of hardwoods and softwoods (mixedwood forest). Under *resilience and transition* management black birch, red oak, and white oak is planted following harvest
Group selection	GroupSelection3	0.4 - 0.6 hectare gaps created on a 20-year cutting cycle. Located on sites where beech is more prevalent. Located primarily within northern hardwood and mixedwood sites. Treatment areas are planted post-harvest with bitternut and shagbark hickory, red oak, and black cherry.
Intermediate thinning	Thinning	30-40% of overstory removed at each entry. Applied to conifer plantations and stands dominated by common conifers such as eastern white pine, red pine, and hemlock.
Overstory removal	ConiferConversion	Complete overstory removal of oldest cohorts leaving young cohorts behind. Applied to plantation and stands dominated by conifers with adequate advance regeneration present.
Patch Cut/Patch Clearcut	PatchCut	Complete removal in 1-4 hectare patches. Occur on a wide range of sites with the goal of regenerating a mix primarily mid-tolerant and intolerant species
Patch Cut/Patch Clearcut	PatchCut2	Complete removal in 2-4 hectare patches. Occur on a wide range of sites with the goal of regenerating a mix primarily mid-tolerant and intolerant species. Under transitional management treated areas are planted with climate suitable tree seedlings; red oak, bitternut and shagbark hickory, black cherry, and black birch
Shelterwood	Shelterwood	40% removal across mature age classes upon first entry followed by an overstory removal cutting (80-90%) 25-years following the establishment cut.

Harvesting regimes have been shown to be influenced by social factors (Kittredge et al. 2003), but the factors which determine landowners willingness to incorporate adaptive management approach remains unclear. Integration of adaptive planning measures and tactics are expected to increase as climate impacts become more relevant and ownership appears to be a major factor in the acceptance and application of these measures (Ontl et al. 2018). Therefore, we have made assumptions as to the level of application of measures such as climate suitable planting within each ownership type (Table 7).

Table 7: Percent of forest land area treated annually for each ownership and across all management scenarios. Silvicultural treatment shown as LANDIS-II harvest implementation code as described in Table 13. Percent annual climate suitable planting is denoted by *. "Federal – AT" ownership is managed to with no harvesting

Ownership	Silvicultural Treatment	Current management	Resistance and Resilience	Resilience and Transition
Private	*SingleTree*	1.00%	0.20%	0.04%
	Partial	0.20%	0.05%	0.01%
	GroupSelection1	0.10%	0.10%	0.20%
	GroupSelection2	0.05%	0.10%	0.33% (0.001%*)
	GroupSelection3	0.00%	0.20%	0.83% (0.001%*)
	Thinning	0.20%	0.20%	0.20%
	PlantationConversion	0.05%	0.10%	0.10%
	PatchCut	0.05%	0.12%	0.00%
	PatchCut2	0.00%	0.30%	0.10% (0.0001%*)
	Shelterwood	0.02%	0.03%	0.03%
	TOTAL	***1.67%***	***1.45%***	1.84%
Public	*SingleTree*	0.80%	0.10%	0.10%
	GroupSelection1	0.40%	0.10%	0.20%
	GroupSelection2	0.20%	0.10%	0.50% (0.001%*)
	GroupSelection3	0.00%	0.40%	0.70% (0.001%*)
	Thinning	0.40%	0.40%	0.20%
	PlantationConversion	0.10%	0.10%	0.15%
	PatchCut	0.10%	0.10%	0.30%
	PatchCut2	0.10%	0.10%	0.10% (0.0001%*)
	Shelterwood	0.02%	0.05%	0.02%
	TOTAL	***1.22%***	***1.45%***	***2.27%***
Federal - MABI	*SingleTree*	1.00%	0.20%	0.06%
	GroupSelection1	0.10%	0.10%	0.20%
	GroupSelection2	0.00%	0.20%	0.30% (0.001%*)
	GroupSelection3	0.00%	0.10%	0.10% (0.001%*)
	Thinning	0.50%	0.40%	0.20%
	PlantationConversion	0.20%	0.05%	0.30%
	PatchCut	0.00%	0.10%	0.30%
	PatchCut2	0.00%	0.00%	0.00%
	Shelterwood	0.00%	0.00%	0.00%
	TOTAL	***1.8%***	***1.15%***	***1.36%***
Federal - AT	*No Harvest Management*	*NA*	*NA*	*NA*

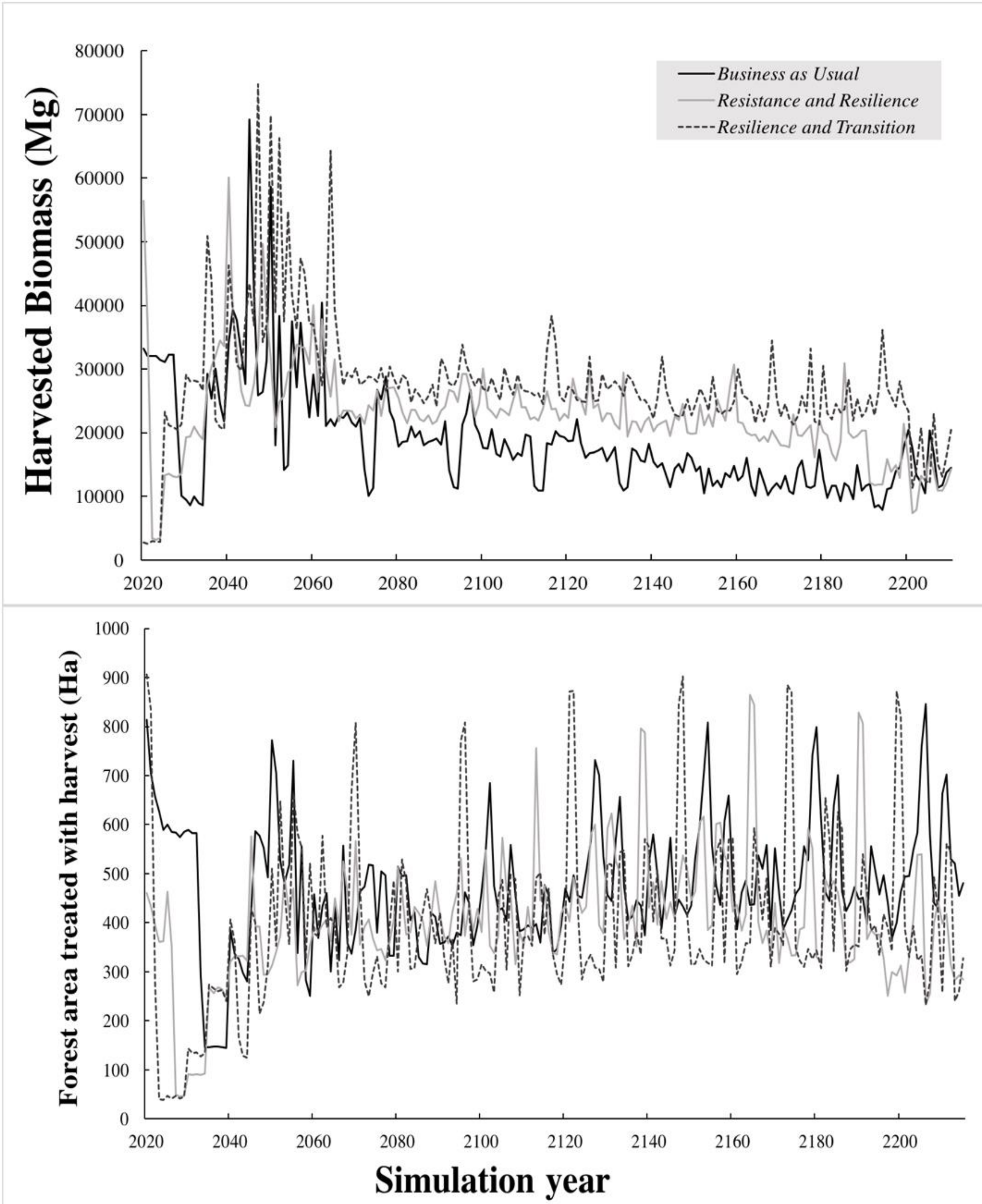

Figure 5: Simulated total annual harvest for each management regime (Top) and forest area treated annually with harvest (Bottom).

To assess the impacts of the application of adaptive tactics on future forest composition, we developed two alternative management regimes. The first alternative, which we refer to as *Resistance and Resilience (RR)*, is designed to promote an increased diversity of tree species, age classes, and adaptive responses among tree species to projected climate changes. To explore these objectives, we simulate a greater application of group selection and patch selection harvests which create larger openings (≥0.1 ha) than are observed currently within the landscape to allow for tree species with wider range of shade tolerances to establish (in comparison to single tree selection which favors shade-tolerant species) (Table 7). Age class diversity is promoted by applying these harvest treatments on a semi-regular interval (cutting cycle) within the same stand, referred to as uneven-aged management.

The second alternative management regime, which we refer to *Resilience and Transition*, aims to increase diversity of tree species, age classes, and adaptive responses present within forested stands in the future but is differentiated from the *Resistance and Resilience (RT)* regime in the level of landowner application of measures which assist the transition of forest systems. To simulate these objectives, we employ a similar approach with greater application of uneven-aged management using group selection harvests but include a greater representation of gaps sizes > 0.2 ha. Red maple, yellow birch, black birch, black cherry, and red oak are projected to maintain suitable habitat under future climate scenarios and were therefore not harvested at the same intensity as was done in the other two management scenarios. Adaptive response diversity was promoted through enrichment planting of future adapted/climate suitable tree seedlings in recently harvested areas. (Table 7).

Annual harvest rate under the *business as usual* (*BAU)* management scenario was 18,606 Mg·yr^{-1} for the 200-year simulation. The annual harvest rate for *RR* and *RT* were 22,870 Mg·yr^{-1} and 27,646 Mg·yr^{-1} respectively (Table 7, Figure 5). Mean forest area harvested varied between management scenarios with *BAU* management treating the largest area (Table 8). Under *BAU* management a larger area was treated annually, however the volume removed annually was the lowest. This is due in part to the greater application of partial harvesting, such as single-tree selection, which removed lower amount of above ground biomass per unit area in comparison to group selection harvests, which were adopted more broadly under *RR* and *RT* management.

Table 8: Simulated annual harvest intensity for three simulated management scenarios.

	Mean harvest volume (Mg·ha^{-1}± *SD*)	**Mean forest area harvested (Mg·ha^{-1}± *SD*)**	**Percent landscape harvested**
Business as Usual (BAU)	18,605 ± *8,702*)	475 ± *128*	1.01%
Resistance and Resilience (RR)	22,870 ± *7,514*)	405 ± *131*	0.86%
Resilience and Transition (RT)	27,646 ± *9,591*)	394 ± *175*	0.84%

2.2.6 Data Analysis

To assess tree species compositional shifts, we summarized changes in individual tree species above ground biomass. The biomass succession extension in Landis-II provides total above ground biomass and individual tree species total biomass output at designated time steps. We utilized these outputs to summarize change in above ground biomass and relative above ground biomass (relative biomass=individual tree biomass/total above ground biomass), which we expressed as percentage (%) for each individual tree species for every year of the two-hundred-year simulation.

Visual representation of current and forest composition (at year 2110 and 2210) are presented. We utilize the biomass reclassification output (v3.0) extension for Landis-II, which reclassifies sites within the landscape into user specified forest types based on species groupings. The forest types are dependent on species dominance (total above ground biomass) at that site. The result from the process allows for visual representation of forest type changes over time. We present forest type classification at the beginning of the simulation and for year 2110 and 2210.

To examine species compositional gradients as they relate to nine climate-management scenarios at two time steps (2110 and 2210), we utilized a non-metric multidimensional scaling (NMS) ordination on projected species relative biomass values (PC-ORD Version 7.0; McCune and Mefford, 2011). A general relativization was applied to equalize the individual species contribution to the ordination solution. The resulting ordination was graphed showing the two axes which explained the highest percentage of the variance. Individual tree species' relative biomass values which had a significant contribution to the ordination solution, based on Kendall's tau (τ), were shown in ordination space for further examination.

The effects of climate and management on future biomass conditions were examined using mixed model analysis of variance (ANOVA) in R (R-Team, 2015). Total landscape above ground biomass (AGB) and total AGB for four focal species was averaged across four replicates of each of the nine climate-management scenarios at the final year of the simulation (year 2210). Focal species analyzed were sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*), eastern hemlock (*Tsuga canadensis*), and red spruce (*Picea rubens*). Sugar maple, American beech, and eastern hemlock were analyzed because they comprised the highest proportion of biomass at the beginning of the simulation. Red spruce was included in the analysis given its ecological importance to the region. When a significant effect was detected, post-hoc Tukey's honestly significant difference (Tukey HSD) pairwise analysis was used to identify differences between climate-management scenario combinations as it related to differences in total and individual focal species absolute biomass in the final year of the simulations. For all tests an alpha (α) of 0.05 was used.

2.3 RESULTS

2.3.1 Forest composition

Sugar maple (*Acer saccharum),* American beech (*Fagus grandifolia),* and eastern hemlock (*Tsuga canadensis)* are currently the three most abundant species based on absolute and relative above ground biomass (AGB) within the landscape (Figure 6). Sugar maple occurs in high abundance throughout the landscape with American beech occurring in highest abundances in the western portion of the landscape at slightly higher elevations. Eastern hemlock dominates in much of the eastern portion of the landscape and is commonly found on the steeper upper slopes along waterways and drainages. White pine (*Pinus strobus),* red spruce (*Picea rubens),* white ash (*Fraxinus americana)* and yellow birch (*Betula allegeniensis)* all occur at lower abundances based on relative AGB but are found commonly throughout the landscape.

Sugar maple is projected to increase in relative biomass over the next century regardless of climate scenario (Figure 8); however, relative AGB declined under current climate conditions over the next two centuries and increased under RCP 4.5 and RCP 8.5 scenarios during this time period (Figure 8, 9, 10). Similarly, American beech is projected to increase in relative biomass over the next century with relative AGB declining by the end of the next two centuries under current climate but increasing under both RCP 4.5 and RCP 8.5 climate scenarios (Figure 8,9,10). Eastern hemlock relative biomass is projected to increase slightly under both climate scenarios but increase rapidly by the end of the next two centuries under current climate conditions (Figure 7,8,9,10). Red spruce is projected to increases in relative biomass under current climate but a changing climate (RCP 4.5 & RCP 8.5) is projected to limit the AGB of this species (Figure 10)

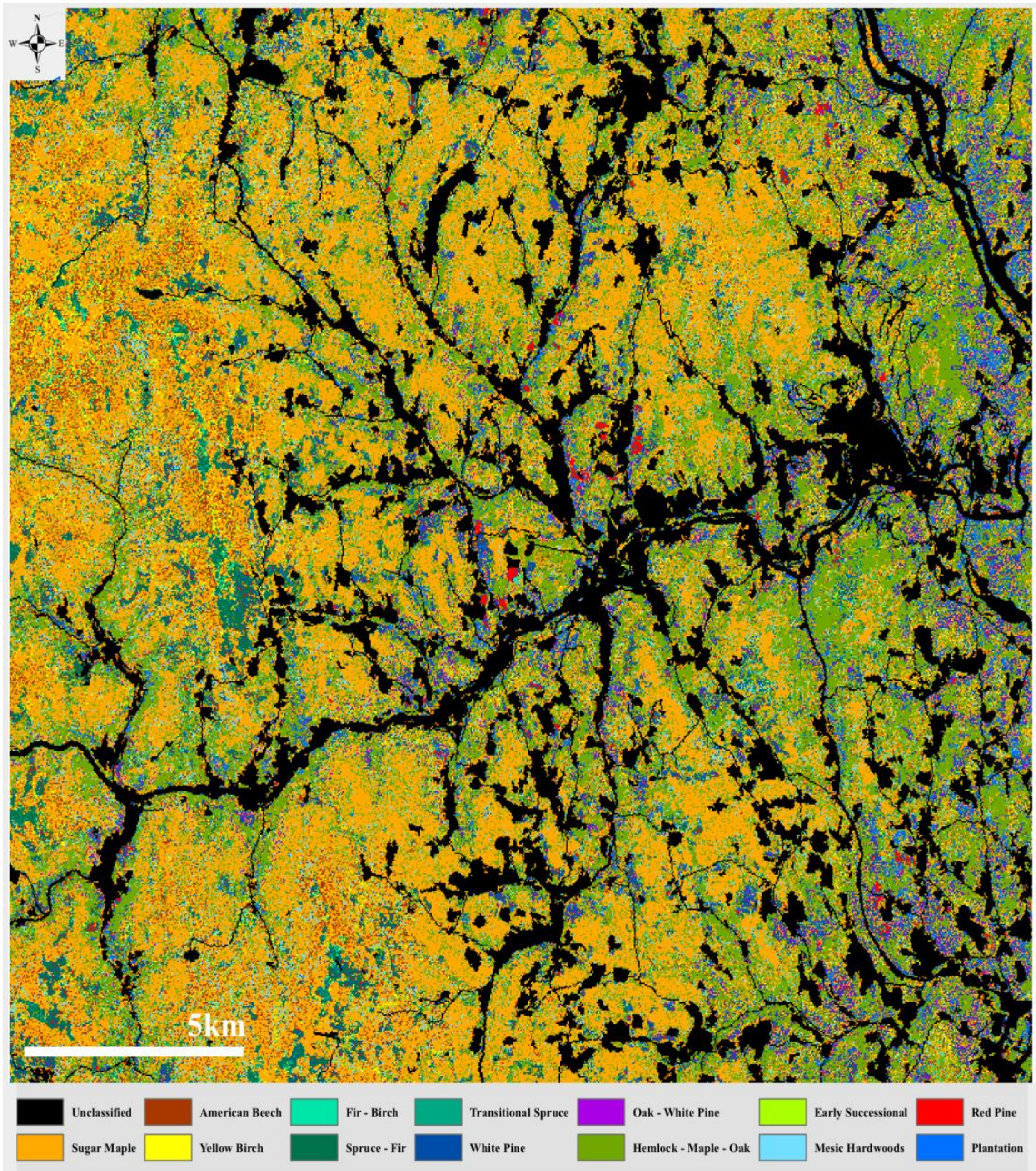

Figure 6: Map of initial forest community type based on dominant species within each 30x30 meter cell. Sugar maple, beech, yellow birch, white pine, hemlock (*Tsuga canadensis),* Norway spruce, and red pine are all dominated by the single species. The other forest types are comprised of tree species groups. Fir – Birch: *Abies balsamea* and *Betula papyrifera*; Spruce – Fir: *Picea rubens, Abies balsamea* and *Betula papyrifera*; Transitional Spruce: *Picea rubens, Betula alleghaniensis,* and *Betula papyrifera*; Oak – Pine: *Quercus rubra, Quercus alba, Pinus strobus, Carya ovata, Carya cordiformis*; Hemlock – Hardwood: *Quercus rubra, Quercus alba, Acer rubrum, Tsuga canadensis*; Early successional: *Prunus pensylvanica, Prunus serotina, Betula papyrifera, Populus grandidentata, Populus tremuloides*; Mesic – Hardwood: *Fraxinus americana, Tilia americana, Quercus alba, Betula lenta;* Plantation species: *Pinus sylvestris and Larix decidua*. Cells are reclassified to show the forest type with the highest total biomassreclassified to show the forest type with the highest total biomass

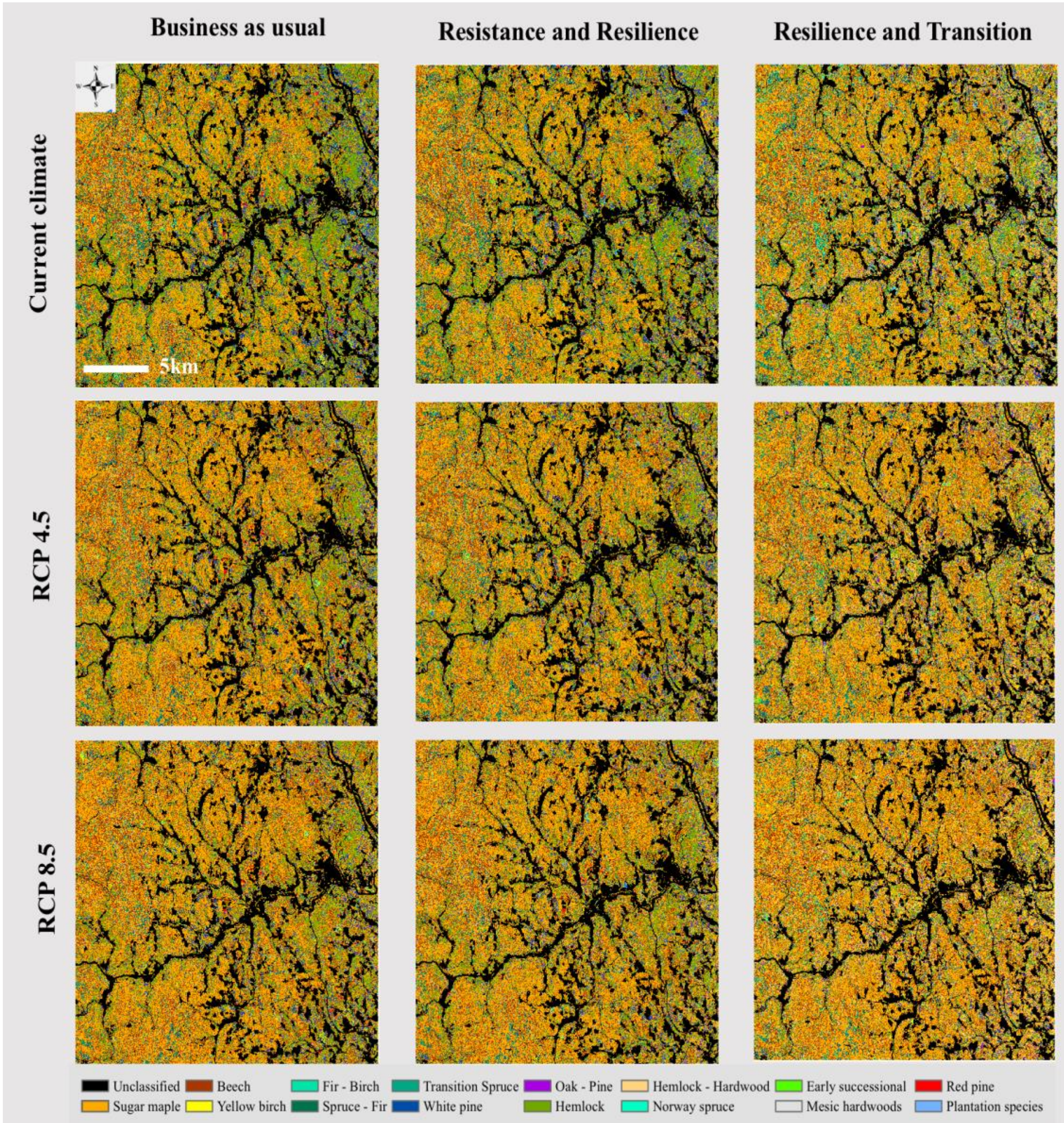

Figure 7 Map of forest composition at year 2110 under nine climate and management scenarios

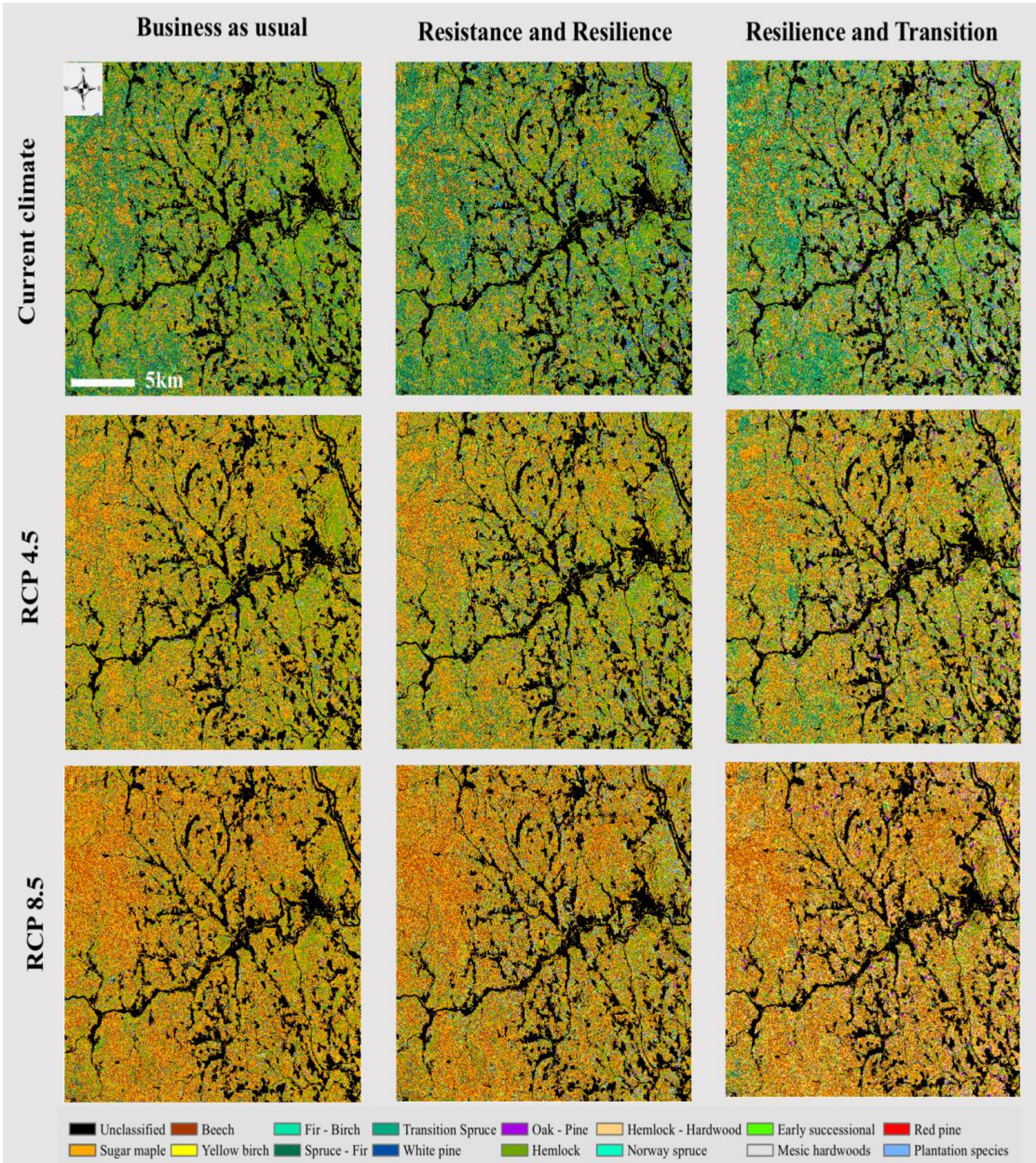

Figure 8: Map of forest composition at year 2210 under nine climate and management scenarios

Simulated management alternatives had minimal impacts on future forest composition within the landscape. While the relative biomass of the dominant tree species was not altered by alternative management by the end of the simulation period, several species with a lower proportion of live tree biomass at the beginning of the simulation were projected to increase in relative biomass as a result of alternative management. Under RR and RT, red maple (*Acer rubrum)* is projected to increase by year 2210 under RCP 8.5 (Appendix I). Early successional species such as bigtooth aspen (*Populus grandidentata),* quaking aspen (*Populus tremuloides),* and paper birch (*Betula papyrifera)* are projected to increase modestly in relative biomass under RR and RT treatments. It is projected that paper birch (*Betula papyrifera)* will increase under current climate but not under RCP 4.5 or RCP 8.5 (Appendix I). Red oak (*Quercus rubra)* is projected to increase in relative biomass under RR treatment by the final year of the simulation, especially under the RCP 8.5 (Appendix I). Yellow birch is also expected to respond favorably to RR and RT treatments, most likely due to the larger harvest gap size promoted under these treatments, which favors species with moderate shade tolerance. Yellow birch did best (i.e. largest increase in relative biomass) under current climate and RCP 4.5, respectively (Appendix I). RT management, as expected, did increase proportional biomass of oak (*Quercus spp.)* and hickory (*Carya spp.)* due to planting and an intentional reduction in harvest rates of these species.

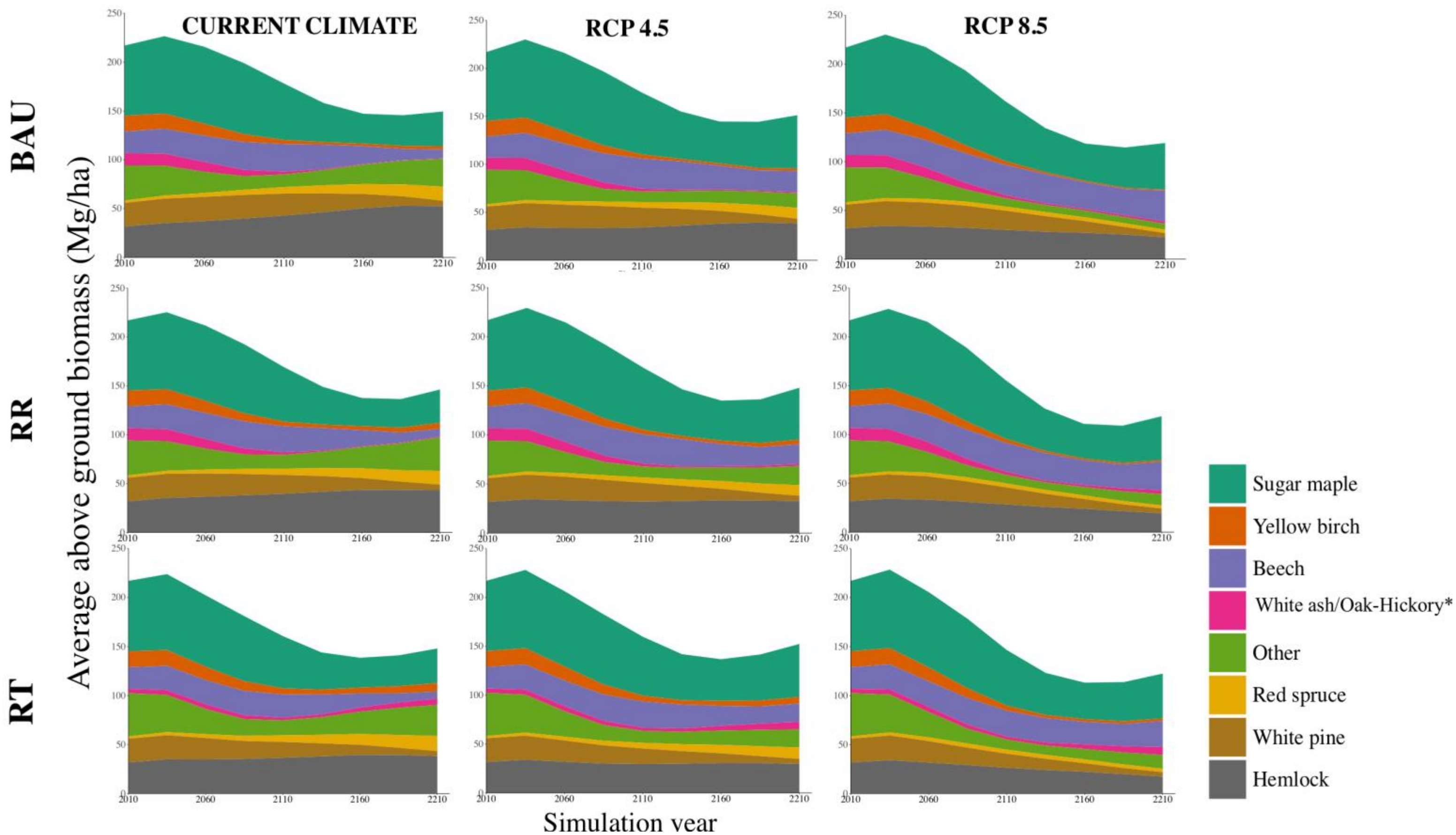

Figure 9: Projected change tree species average total above ground biomass (AGB) (Mg/ha) from year 2010 to year 2210 for nine climate/management scenarios. Management approach is shown along the y-axis and denoted by BAU = *Business as usual*, RR = *Resistance and Resilience*, and RT = *Resilience and Transition.* The six species with the highest initial average AGB are shown (*Acer saccharum, Betula allegeniensis, Fagus grandifolia, Fraxinus americana, Pinus strobus, and Tsuga canadensis). Picea rubens* is included due to it ecological significance. For the RT (*Resistance and Transition)* management scenario, White ash is included in the "other" category and Oak-Hickory* (*Quercus rubra*, *Quercus alba*, *Carya ovata*, *Carya cordiformis*) is included given these tree species are promoted through reduced harvesting and are planted as an adaptation strategy. "Other" is comprised of the tree species found in relative low abundance

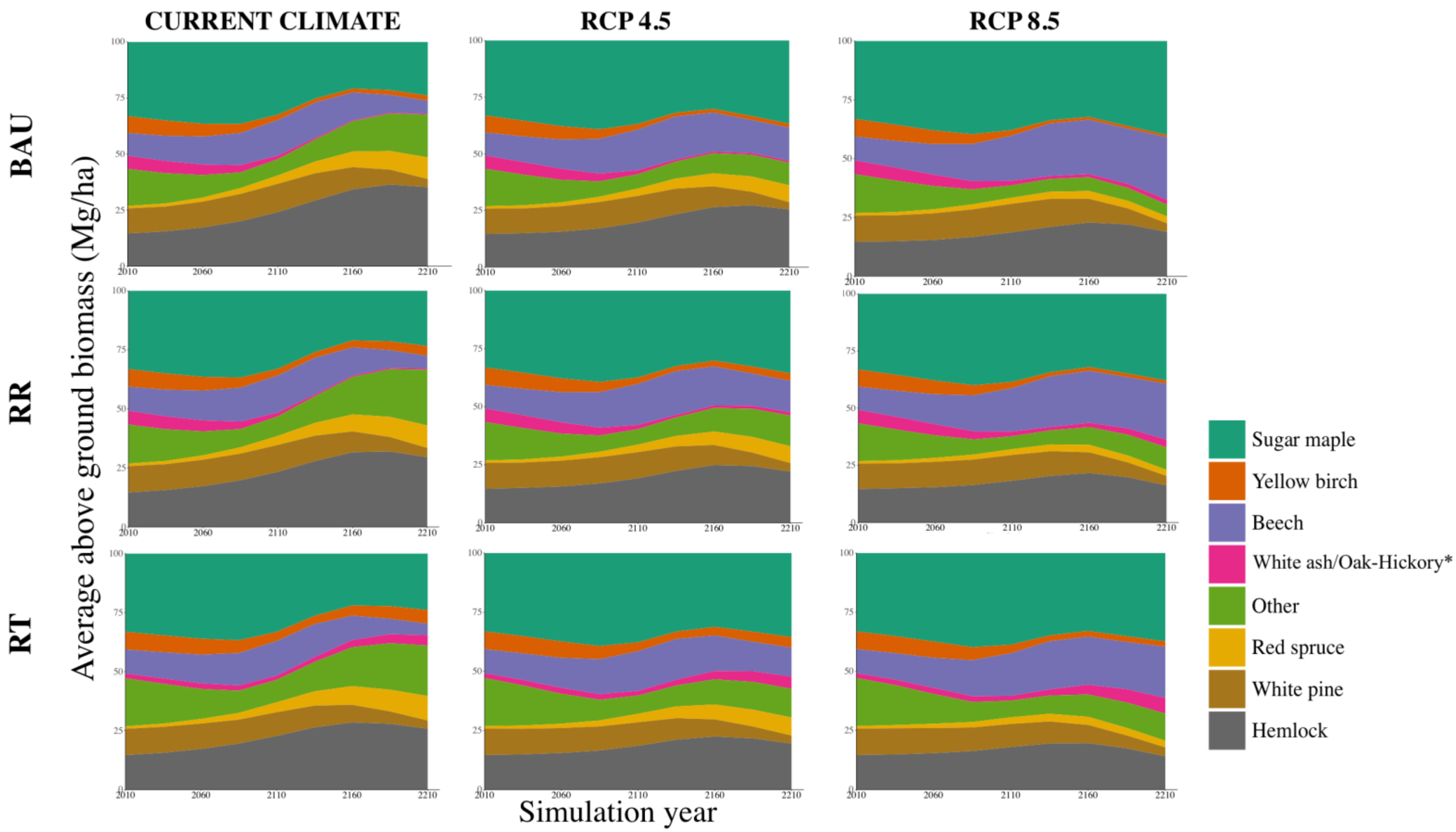

Figure10: Projected change tree species average relative above ground biomass (AGB) (Mg/ha) from year 2010 to year 2210 for nine climate/management scenarios. Management approach is shown along the y-axis and denoted by BAU = *Business as usual*, RR = *Resistance and Resilience*, and RT = *Resilience and Transition*. The six species with the highest initial average AGB are shown (*Acer saccharum*, *Betula allegeniensis*, *Fagus grandifolia*, *Fraxinus americana*, *Pinus strobus*, *and Tsuga canadensis). Picea rubens* is included due to it ecological significance. For the RT (*Resistance and Transition)* management scenario, White ash is included in the "other" category and Oak-Hickory* (*Quercus rubra*, *Quercus alba*, *Carya ovata*, *Carya cordiformis*) is included given these tree species are promoted through reduced harvesting and are planted as an adaptation strategy. "Other" is comprised of the tree species found in relative low abundance.

Species diversity at the beginning of the simulation was 2.17 based on relative species biomass and Shannon's index of diversity (H'). Species diversity declined overtime in all scenarios with the exception of the RT and Current climate scenario which resulted in increased diversity by the end of the simulation (H'=2.21; Table 9). Species diversity was highest under current climate conditions and lowest under RCP-8.5 climate (Table 9). BAU management resulted in the lowest diversity and RT management promoted the highest levels of diversity regardless of climate scenario (Table 9).

Table 9: Shannon's diversity index as derived from average species biomass for nine climate/management scenarios. Shannon's diversity at the beginning of the simulation was 2.17.

Management/Climate Scenario	**Shannon's Diversity Index**	
	Year 2110	*Year 2210*
Business as usual - Current	1.87	1.91
Resistance and Resilience - Current	1.91	2.10
Resilience and Transition - Current	1.97	2.21
Business as usual - RCP 4.5	1.83	1.80
Resistance and Resilience - RCP 4.5	1.84	1.98
Resilience and Transition - RCP 4.5	1.90	2.08
Business as usual - RCP 8.5	1.79	1.62
Resistance and Resilience - RCP 8.5	1.80	1.81
Resilience and Transition - RCP 8.5	1.86	1.95

The observed differences in future forest composition across a range of management and climate scenarios was supported by a NMS ordination (stress = 2.66, instability = 0.000) which explained 89.7% of the variation in species relative biomass along two axes (Figure 11). Axis 1 captured 53.4% of the variation and was positively related to relative AGB of eastern white pine ($\tau = 0.582$, $p < 0.000$) and red pine ($\tau = 0.595$, $p < 0.000$) and largely contained scenarios in the year 2110. Axis 1 was negatively related to relative AGB of bigtooth aspen ($\tau = -0.503$, $p = 0.004$), quaking aspen ($\tau = -0.582$, $p < 0.000$), bitternut hickory ($\tau = -0.529$, $p = 0.006$), shagbark hickory ($\tau = -0.511$, $p = 0.006$), white oak ($\tau = -0.598$, $p = 0.002$), and red oak ($\tau = -0.778$, $p < 0.000$) and contained scenarios in the year 2210. Axis 2 represented 36.3% of the variation and was positively related to relative biomass of balsam fir ($\tau = 0.882$, $p < 0.000$), striped maple ($\tau = 0.582$, $p < 0.000$), Norway spruce ($\tau = 0.765$, $p < 0.000$), red spruce ($\tau = 0.686$, $p < 0.000$), eastern hemlock ($\tau = 0.856$, $p < 0.000$), and scots pine ($\tau = 0.699$, $p < 0.000$) with scenarios associated with current climate conditions in this portion of the axis. Axis 2 was negatively related to relative AGB of white ash ($\tau = -0.869$, $p < 0.000$), American beech ($\tau = -0.725$, $p < 0.000$), red maple ($\tau = -0.712$, $p < 0.000$), sugar maple ($\tau = -0.699$, $p < 0.000$), and hophornbeam ($\tau = -0.569$, $p = 0.001$) and was associated with RCP 4.5 and 8.5 scenarios.

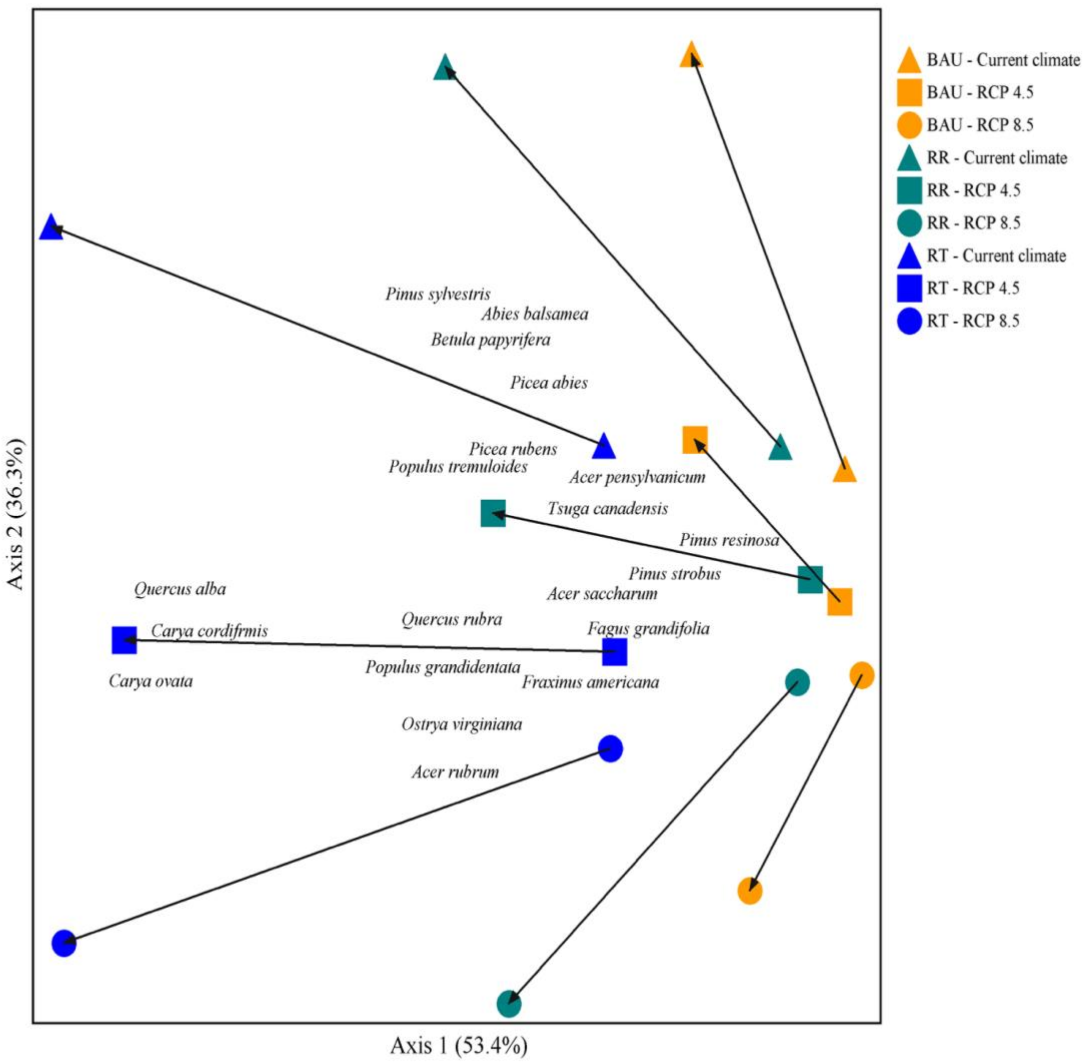

Figure 11: Nonmetric multidimensional scaling (NMDS) ordination of proportional mean species above ground biomass along the two main axes which represent the highest percentage of variation. Species with significant correlations with either axes are shown. Nine climate and management scenarios are shown with color signifying management type and shape signifying climate scenario. Successional vectors are drawn between each climate-management treatment from year 2110 to year 2210.

2.3.2 Aboveground biomass

Aboveground live biomass (AGB) is projected to increase over the next twenty-five years and then decline for the next one-hundred and fifty years across all management and climate scenarios. While biomass trajectories are projected to stabilize by the end of the simulation period,

AGB trajectories show signs of differentiation around year 2110 with RCP 8.5 climate model projections showing steeper declines (Figure 12).

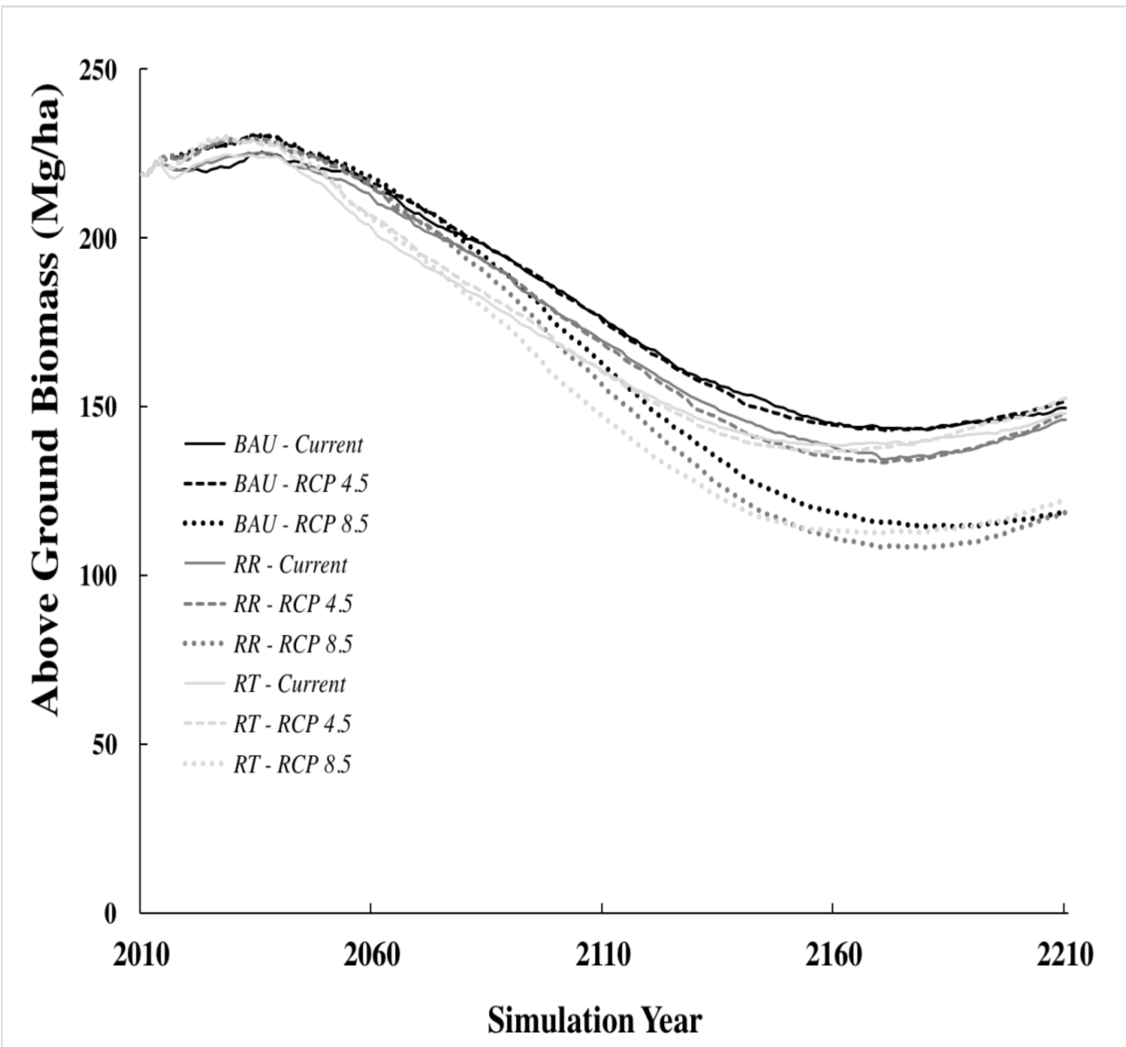

Figure 12: Change in average aboveground biomass $Mg{\cdot}ha^{-1}$ simulated across nine management-climate scenarios for a 200 year period from 2010 – 2210

Aboveground biomass at the final year of the simulation (2210) was influenced by climate but not by management (*$p < 0.001$).* In the final year of the simulations, AGB under current climate (*147.90 ± 0.79 Mg·ha*$^{-1}$) and RCP 4.5 (*150.55 ± 1.04 Mg·ha*$^{-1}$*)* were not significantly different from each other (*$p = 0.77$)* but both had higher levels of AGB than under RCP 8.5 (*120.09 ±4.51 Mg·ha*$^{-1}$*; $p < 0.01$;*Figure 12).

Sugar maple biomass in the final year of the simulation was not influenced by management but was influenced climate (*$p<0.001$),* resulting in higher levels of AGB in both RCP 4. 5 and RCP 8.5 climate projections (Figure 13). Sugar maple biomass in the final year of the simulation under RCP 4.5 (54.11 ±*0.41 Mg·ha*$^{-1}$*)* was significantly higher

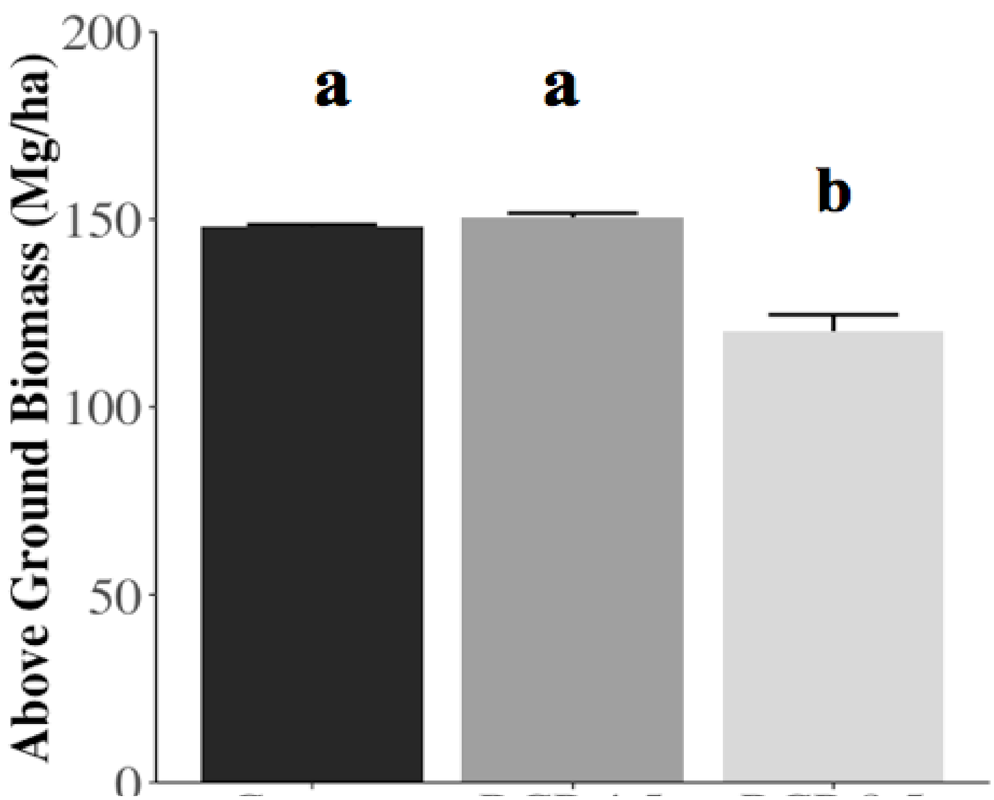

Figure 13: Average aboveground biomass (Mg) at the final year of the simulation (2210). Final simulation year biomass was a function of climate but not a function of management. Lowercase letters indicate significant differences between climate scenarios at $\alpha = 0.05$.

than biomass levels under RCP 8.5 (46*.09 ± 2.80* $Mg \cdot ha^{-1}$*; p=0.004)* and under current climate projections (35*.05 ± 0.34* $Mg \cdot ha^{-1}$*; p<0.001).*

American beech AGB in the final year of the simulation was highest under RCP 8.5 (29.14 *± 0.96* $Mg \cdot ha^{-1}$) in comparison to current climate (7.92 *± 0.21* $Mg \cdot ha^{-1}$*; p < 0.001)* and RCP 4.5 (20.27 *± 1.31* $Mg \cdot ha^{-1}$*; p < 0.001;* Figure 14; Table 10*)*. Eastern hemlock AGB in the final year of the simulation was influenced by climate and management but not their interaction. Eastern hemlock AGB was highest under current climate (44.69 *± 1.83* $Mg \cdot ha^{-1}$*; p<0.001*) in comparison to RCP 4.5 (33.56 *± 1.23* $Mg \cdot ha^{-1}$*; p=0.001*) and RCP 8.5 (19.65 *± 1.75* $Mg \cdot ha^{-1}$*; p<0.000;* Figure 14; Table 10). Eastern hemlock AGB in the final year of the simulation was not significantly different (*p > 0.05)* between management approaches (BAU, RR, RT) when climate was held constant. However, when a comparison was made between management and climate scenarios, BAU management under current climate projections resulted in the highest levels of eastern hemlock AGB in the final year of the simulation when compared to all other management – climate combinations (*p<0.05).* Red spruce AGB was influenced by climate and was highest under current climate (14.57 *± 0.22* $Mg \cdot ha^{-1}$) and significantly lower under RCP 4.5 (11.21 *± 0.57* $Mg \cdot ha^{-1}$*;p<0.001*) and under RCP 8.5 (3.37 *± 0.45* $Mg \cdot ha^{-1}$*;p<0.001;*Figure 14; Table 10) respectively in the final year of the simulation.

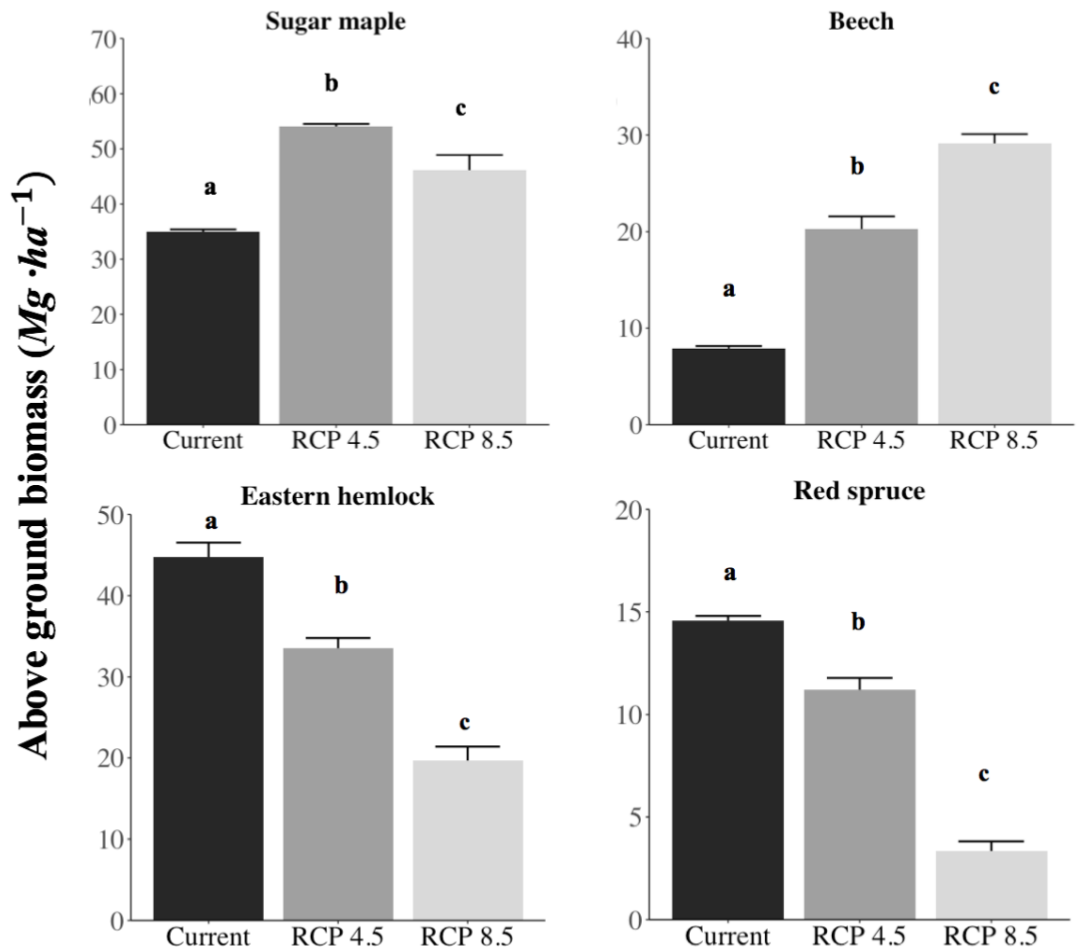

Figure 14: Aboveground biomass in the final year of the simulation (2210) across three climate scenarios for four focal species: Sugar maple (*Acer saccharum);* American beech (*Fagus grandifolia);* eastern hemlock (*Tsuga canadensis*); and red spruce (*Picia rubens).* Lowercase letters indicate significant differences between climate scenarios at $\alpha = 0.05$.

Table 10: F-statistic results from ANOVA showing the influence of climate, management, and their interaction on future biomass condition for four focal species simulated.

		Acer saccharum		*Fagus grandifolia*		*Tsuga canadensis*		*Picea rubens*	
Above ground biomass ($Mg \cdot ha^{-1}$)	df	*F*	*p* value	*F*	*p* value	*F*	*p* value	*F*	*p* value
Climate	2	28.55	<0.001	136.28	<0.001	127.49	<0.001	154.78	<0.001
Management	2	0.406	0.671	3.41	0.047	18.72	<0.001	0.853	0.437
Climate x Management	4	0.02	0.999	0.342	0.85	1.56	0.215	0.142	0.965

2.4 DISCUSSION

2.4.1 Climate change impacts on forest composition and above ground biomass

Numerous factors are predicted to impact the structure and composition of forests in northeastern North America. These impacts will occur in a landscape recovering from centuries of intensive land-use and currently composed of a diversity of landowners associated management objectives, making predictions based on climate impacts alone challenging. This study lends support to a growing body of work highlighting the general inertia in forest conditions over the next century, with impacts manifesting two to three centuries into the future (Duveneck et al. 2017, Iverson et al. 2017, Wang et al. 2017, Janowiak 2018, Liang et al. 2018). Although adaptation strategies increased the representation of future-adapted species, the limited levels of application of these approaches across the diverse ownerships in the region suggest significant challenges to adaptive management strategies in counteracting climate change impacts on forests.

The results of our simulations suggest forest composition is not projected to change dramatically over the next century under multiple climate and management scenarios within the study landscape. However, forest composition in the region is likely to undergo substantial shifts over the next two centuries. Sugar maple, America beech, eastern hemlock, and red spruce, which are all long-lived, shade-tolerant species, are expected to remain stable or gain in relative biomass over the next century. Projected climate change is not expected to shift current composition or successional trajectories in the next century, which is consistent with the findings from similar studies in the region (Duveneck et al. 2017, Wang et al. 2017, Duveneck and Thompson 2019). As forests in our region continue to recover from past intensive land use, the inertia associated with long-term successional dynamics appears to define forest conditions in the near term. Recent work in the region suggests that climate will have an increasingly significant impacts on forest and biomass condition beyond the next century (Duveneck et al. 2017, Iverson et al. 2017, Wang et al. 2017). Our study corroborates these findings and suggests that climate will significantly influence forest composition and biomass conditions over the next two centuries in our study region.

Future forest conditions assuming a continuation of current climate conditions largely reflected long-term successional dynamics and recovery from historic land use. The continuation of current climate is likely to promote the continued successional progression of eastern hemlock resulting in large increases in relative biomass over the next two centuries. Our study did not simulate the potential impacts of hemlock woolly adelgid which is predicted to have large impacts on the future abundance of eastern hemlock in the region (Dukes et al. 2009). The likely recovery of red spruce relative biomass is consistent with recent work showing improved growth rates for the species, likely linked to reduced acid-deposition and favorable growing conditions (Kosiba et al. 2018), and an expansion of montane spruce-fir communities down slope as these species appear to be recovering from historical selective harvesting (Foster and D'Amato 2015). However, as the climate in the region shifts from historical norms, eastern hemlock, red spruce, and other northern temperate species (i.e. balsam fir and paper birch) show an increasing sensitivity to a warming temperatures and reduction in recruitment rates due to rising evaporative demand among other physiological stressors associated with a changing climate, which is consistent with previous findings (Iverson et al. 2008, Iverson et al. 2017, Janowiak 2018).

Shifts in forest community composition are likely to occur along climate gradients, disproportionately favoring southern species with lower climate sensitivity (Iverson et al. 2017, Janowiak 2018). Our study supports these predictions with species such as red spruce, paper birch, and balsam fir all expressing a sensitivity to warming climate conditions. Red oak and red maple tended to favor better under a warming climate likely related to their relatively lower climate sensitivity. This study suggests that within our study region, sugar maple and American beech are likely to express lower climate sensitivity over the next two centuries. Climate is projected to have a positive impact on sugar maple and American beech relative biomass within the region in comparison to other species commonly associated with northern hardwoods (i.e. yellow birch and red spruce).

The projected expression of lower climate sensitivity by sugar maple is consistent with regional modeling efforts which suggest under moderate warming projections sugar maple might not see major reductions in realized habitat (Iverson et al. 2017, Janowiak 2018). In contrast, recent work by Oswald et al. (2018) suggests that sugar maple health and signs of decline regionally is related to climatic variables and a suite of interacting stressors. Recent work by Bose et al. (2017) indicate that sugar maple is declining in absolute and relative abundance across the northeast while American beech is increasing. However, sugar maple does remain competitive on sites where abiotic factors (soil productivity) are favorable to sugar maple. The study area for this project does contains rich and productive soils when compared to soils in other regions across the northeast, which may mediate climate related sugar maple decline observed elsewhere. In addition, findings from Bose et al. (2017) suggest that our study area is located within an area where American beech is not increasing at a rate as high as other regions. While site quality may mediate the sugar maple decline observed in other regions, our study does not simulate the interacting and often compounding nature (i.e. drought stress preceded by repeated insect defoliation etc.) of environmental stressors impacting tree species growth, development, and establishment/recruitment and therefore, may underrepresent sugar maple's climate sensitivity in the region. In addition, the model does not account limited regeneration success resulting in part due to increased competition from prolific American beech sprouting. However, our model results may indicate that in the absence of major compounding stressors, under low intensity disturbance, and with favorable site conditions, sugar maple is likely to remain a dominant species within the region.

As forests in the region continue to recover from past intensive land use around 150 years ago, average biomass accumulation is expected to peak as forest reach maturity (Bormann 1979, Halpin and Lorimer 2016). Maximum levels of biomass accumulation vary across the landscape and are determined by multiple factors, including but not limited to, stand age, site conditions, and disturbance histories (Keeton et al. 2011, Halpin and Lorimer 2016, D'Amato et al. 2017a). While forest cover has declined in the northeast over the past decade due to forest conversion (Foster 2010) average biomass is expected to continue to accrue for the next fifty years (Duveneck et al. 2017, Duveneck and Thompson 2019). In support of these findings our study projects that average biomass will likely continue to increase over the next three decades within the study region. Peak average above ground biomass projected for the landscape (approximately 230 Mg/ha) is within the range of observed biomass levels recorded within managed secondary temperate forest stands in the region (Urbano and Keeton 2017) and within the range of projected biomass trajectories under simulated natural disturbance regimes (Halpin and Lorimer 2016, Duveneck et al. 2017).

Our study projects biomass development trajectories to decline through the end of this century and into the middle of the next century before stabilizing. These trends are expected within the region as forests mature to the point where factors attributed to demographic and stand dynamics begin to alter the trajectory of average live biomass accumulation. We would also expect to see similar trends in biomass trajectories within a landscape where frequent, but low intensity forest harvesting is occurring and where natural disturbances and site level factors are also contributing to variable rates of biomass development across the landscape. These trends are consistent with the extensive field and simulation work conducted by Halpin and Lorimer (2016) who expect average aboveground live-tree biomass to follow a peak-decline-stabilize trajectory similar to that proposed by Bormann and Likens (1979).

Much like the species compositional shifts expected for the study area, average aboveground biomass trajectories are not projected to change due to climate for the next 100 years. However, we do expect climate to play an increasingly significant role resulting in lower biomass levels by the end of the simulation period. Average above ground biomass levels are expected to decline more rapidly and remain at lower levels under more extreme warming conditions. These divergent biomass trajectories under high emissions climate change scenarios is likely due to reduced productivity and establishment of climate sensitive species under protracted climate warming. Higher mortality rates related to greater physiological stress and reduced growth rates under prolonged droughts and extreme temperature conditions are likely also contributing to this observed decline in biomass under the highest emissions scenario. In addition, these divergent trajectories could also be linked to the slow migration of tree species with habitat requirements suitable to future conditions to the region under high emissions climate scenarios (Zhu et al. 2012). Reduction in suitable habitat for common species in the region is expected to be greater under climate projections with the greatest projected warming (Iverson et al. 2008, Iverson et al. 2017) which reflects our model projections that many of the dominant species within the landscape will likely experience steep declines in relative biomass under high emission scenarios and reduced rates of establishment which may lead to steeper average biomass reductions overtime.

2.4.2 Impacts of forest adaptation strategies on future forest conditions

There remains considerable interest in the use of forest adaptation strategies to address the uncertainty of climate change impacts on forests. A range of forest adaptation strategies have been proposed (Millar et al. 2007), simulated (Duveneck and Scheller 2015, 2016), evaluated (D'Amato et al. 2011, 2013) and implemented (Nagel et al. 2017) throughout the northeast. While forest management remains the most prominent forest disturbance agent in the region (Duveneck and Thompson 2019), few studies have evaluated the potential long-term impacts of operationally feasible adaptation strategies. Our study suggests that the simulated level of adaptation strategies applied within the region may not be enough to counteract the shift in forest compositional and biomass trajectories currently being driven by succession and climate.

Forest adaptation strategies simulated in this study promoted increased forest resiliency through an increased application of uneven aged management and harvest treatments which promoted the regeneration of a wider range of tree species through the creation of larger canopy openings than is typically done under current management. In addition, we simulated the small scale planting of hardwood species projected to do well under a warming climate as a transitional adaptation strategy (Millar et al. 2007, Janowiak et al. 2014, Nagel et al. 2017). These adaptation strategies did not significantly alter future species composition or biomass conditions within the region. However, alternative management did promote a greater diversity of tree species in comparison to business as usual (*BAU)* management. Management alternatives focused on promoting increased forest resilience (*RR management*) did result in increased diversity of tree species at the final year of the simulation and promoted slight increases in relative biomass of red oak, red maple, and early successional species such as bigtooth aspen. Increased application of adaptation strategies aimed at promoting a transition in forest communities towards one that may be better suited to projected future forest conditions (*RT management*) did have a similar impact as *RR management* and resulted in the greatest levels of projected species diversity across all climate scenarios. The increase in species diversity and increased representation of climate suitable species such as *Quercus spp.* and *Ovata spp.* is due to increased canopy opening size, retention of mature oaks and hickories present on the landscape, and the planting of these species throughout the landscape. However, the level of application of adaptation strategies simulated in this study did not significantly shift landscape level composition or biomass trajectories. These findings differ from Duveneck et al. (2014) who projected that climate suitable planting could be used to sustain species diversity and biomass conditions within a landscape in the northern Great Lakes region when applied at high levels. Our study simulated a much lower level of application of these adaptation measures, which would be expected in a region where local silvicultural methods rely heavily on natural regeneration and where planting is not a common practice, potentially explaining the limited influence of climate suitable planting on future forest composition and biomass trajectories in the study region.

Harvest rates for all management scenarios remained at or below the expected harvest rate for the region. Given harvest rates and intensities remain low in comparison to other landscapes and ownerships within the region (Duveneck and Thompson 2019), further integration and application of the adaptation strategies simulated in this study may be attainable and necessary within landscapes with similar forest composition and landowner objectives.

2.4.3 Model limitation and uncertainty

These results present a range of potential futures based on an assessment of plausible climate and management scenarios. As is true for all modeling efforts, these results cannot be interpreted as predictions. Socio-ecological systems are highly complex and cannot be represented fully using a single model. The strengths of a process based model like Landis-II is its ability to represent the interaction between known drivers (i.e. climate, disturbance, succession, etc.) and forest composition and biomass levels on a landscape scale. Given the climate models used for this study only provide projection 100 years, the full range of climate variability that might be expected over the next two centuries is likely underrepresented.

There are multiple socio-ecological processes that interact and impact forest communities at varying scales which were not included in the model framework. We did not include land-use and tenure change dynamics which have been shown to have significant influences of future forest conditions in the region (Thompson et al. 2011, Duveneck and Thompson 2019). While alternative forest management scenarios were presented, we did not fully capture the effect of shifting social, economic, and biophysical conditions on harvest rates and intensity which have been shown to be significant predictive factors (Kittredge et al. 2003, Kittredge et al. 2017, Thompson et al. 2017). In addition, the interactions between climate and the multiple stressors at play in the forests of the northeast which include pests, pathogens, invasive insects and diseases, and herbivory to name a few are not fully captured in these simulations.

2.5 CONCLUSION

This study demonstrates that climate is expected to be an important driver of long-term forest compositional and biomass conditions within the landscape. However, successional dynamics will likely play a larger role in forest compositional shifts over the next century. Forests of the northeast are developing under small-scale disturbance regime and low intensity forest management and favorable moisture and temperature conditions for their recruitment, which together have promoted the establishment and dominance of long-lived shade tolerant species. In the absence of large scale disturbance or dramatic alterations to forest management regimes, current forest composition remains largely unaffected for the next 100 years. In contrast, over the course of the next two centuries climate is likely to play a much more important role in shaping the composition and condition of our forests. Unfortunately, these shifts will likely result in forests which are less diverse as climate sensitive northern temperate species experience major reductions in relative biomass. Under the more extreme warming scenarios, average biomass at the landscape scale is expected to trend significantly lower in comparison to projections under current climate and modest warming.

Reduced species diversity and lower projected biomass levels may impact the provisioning of critical forest based ecosystem services in the future. The projected delayed impacts of climate change on our forests presents an opportunity for resource managers to potentially intervene before the onset of major compositional and biomass shifts occur. Our results suggest that the scale of these interventions may need to be greater than current applications of management given the limited impacts observed under the adaptation scenarios simulated in this study.

CHAPTER 3: MANAGEMENT IMPLICATIONS AND FUTURE DIRECTIONS

3.1 MANAGEMENT IMPLICATIONS

The practice of forestry is one that must rely on the integration of both social and ecological systems. Forest managers act as the intermediary between social expectations and demands on forest goods and services and the actual functionality of the forest ecosystem. Put more simply, foresters manage people as much as they manage forests. Managing forests in the northeast is complex in part because this region is one of the most densely populated regions in the nation and one of the most forested with the predominant forest ownerships type being family forest owners.

Forestry is a unique profession in the sense that managers are constantly considering the long-term impacts of their actions. The realities of managing a system comprised of long-lived species limits the manager's ability to rapidly evaluate the future impacts of management actions or the implications of disturbances. While there remains no crystal ball in which to see the future, managers can and often do rely on the rich ecological knowledge and experience of veteran practitioners. In addition to the reliance of past experience, there has been a committed effort to study forest ecology and the impacts of forest management on future forest conditions with many long-term silvicultural experiments still active in the region, most notably the National Experimental Forest network (D'Amato et al. 2011). While the lived experience of practicing foresters and insights from long-term silvicultural studies remain most critical in guiding management today, the unprecedented levels of change expected in the region may require managers to seek additional guidance from emerging frameworks for addressing these changes.

Recently, managers have increasing begun integrating forest adaptation strategies into management planning. These strategies are often catered to local ecological conditions and objectives through assessments of ecosystem vulnerability and development of silvicultural tactics to reduce climate and forest health impacts (Swanton et al. 2016). Additionally, advances in technology allow researchers to use computer models to simulate future forest growth and change under a range of climate, management, and disturbance scenarios and are often used to provide decision support regarding how to address future changes in environmental conditions. The ability to evaluate a range of potential future conditions represent another tool forest managers can reference when making decisions.

Our study on the long-term impacts of current and alternative forest management and climate change on future forest conditions supports a growing body of knowledge that suggest that climate will have a major impact on future forest condition in the region (Iverson et al. 2008, Duveneck and Scheller 2016, Duveneck et al. 2017, Iverson et al. 2017, Wang et al. 2017, Janowiak 2018). Our work suggests that a changing climate will likely limit the number of species currently inhabiting our forests and favor species better suited to warmer conditions. Our work also suggest that climate induced changes to species composition may take up to a century before they are realized. The past land use history and a relatively low intensity harvesting regime typical in a region dominated by private family forest owners, appears to have set in motion a forest successional trajectory which, in the absence of significant disturbance, will continue to define future forest condition in the near future.

The delayed nature of climate induced shifts in species composition presents an opportunity for resource managers to potentially intervene early and act in ways which might favor assemblages of species projected do well in the future. The inertia of succession is powerful and our study shows that minor modification to management might not be enough to shift future species compositions to a mix of species better suited to future conditions.

The complexity of a landscape comprised on many different and unique ownership objectives presents a challenge in achieving forest adaptation goals at a landscape scale. Widespread application of adaptation tactics may not be attainable, however stand or parcel level interventions may result in localized pockets of sustained diversity and/or increased representation of tree species better suited for future climate conditions.

3.2 STUDY LIMITATIONS AND FUTURE DIRECTIONS

All modeling efforts must be viewed as projections and not predictions. Based on our best understanding of how forest ecosystems function and respond to changing climate and disturbance, we can project plausible future forest conditions. These results present broad trends and general tendencies within a forest system and not the actual future conditions.

There are many interactions that occur within forest ecosystems that influence forest growth and development which cannot be represented fully in this model. While Landis-II, the model used in this research, relies heavily on individual tree species attributes to represent the varied response to disturbance and environmental change, there are species-specific interactions that were not captured in this study. For example, beech bark disease has had dramatic effect on American beech growth, mortality, and recruitment within the region and although our initial forest conditions reflected this, the interactions of this disease complex are not fully represented in the model.

While we did simulate wind disturbance and insect defoliation we could not fully represent the range of forest disturbances that occur within the region. There are a number of forest pests and pathogens which impact forests and often these disturbance agents interact with climate. In addition, land use change and the impacts of deer herbivory, which are significant agents of disturbance in the region, were not simulated in this study.

Finally, there remains some limitations in projecting climatic conditions beyond the next 100 years. The climate models used in this study only represent projected climate variation for the next 100 years and therefore, future projections beyond this time period relies on the continuation of projected trends. In the Landis-II model, a changing climate influences the probability a tree will establish when growing space is made available. Therefore, our ability to represent the full range of climate impacts on tree species establishment beyond 100 years is limited.

The power of these modeling efforts is the ability to make continued adjustments and explore a range of potential scenarios. Future research could focus on the potential influence of landowner decisions in regards to the level of application of adaptive strategies on future forest composition. We know that adaptation strategies are applied at varying rates across the state, and by different types of landowners; however, it remains unclear whether or not these localized applications of adaptation strategies can shift forest composition beyond successional trajectories and towards conditions better suited for projected future climate.

In addition, future studies could explore varied harvest rates and intensities. All three management scenarios in our study simulate harvest rate at or below current harvest rates and the intensity of individual harvests were low. Low intensity harvests are typical for the region and therefore our study may accurately represent long-term impacts of current management but potentially underrepresent the range of potential alternative management approaches.

Finally, as recent research experiments and field trials of adaptation continue to evolve, initial outcomes and approaches suggested by these studies can be integrated into modeling frameworks to better represent how adaptation is being operationalized in the region.

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APENDIX

Appendix I: Individual species projected total and percent biomass

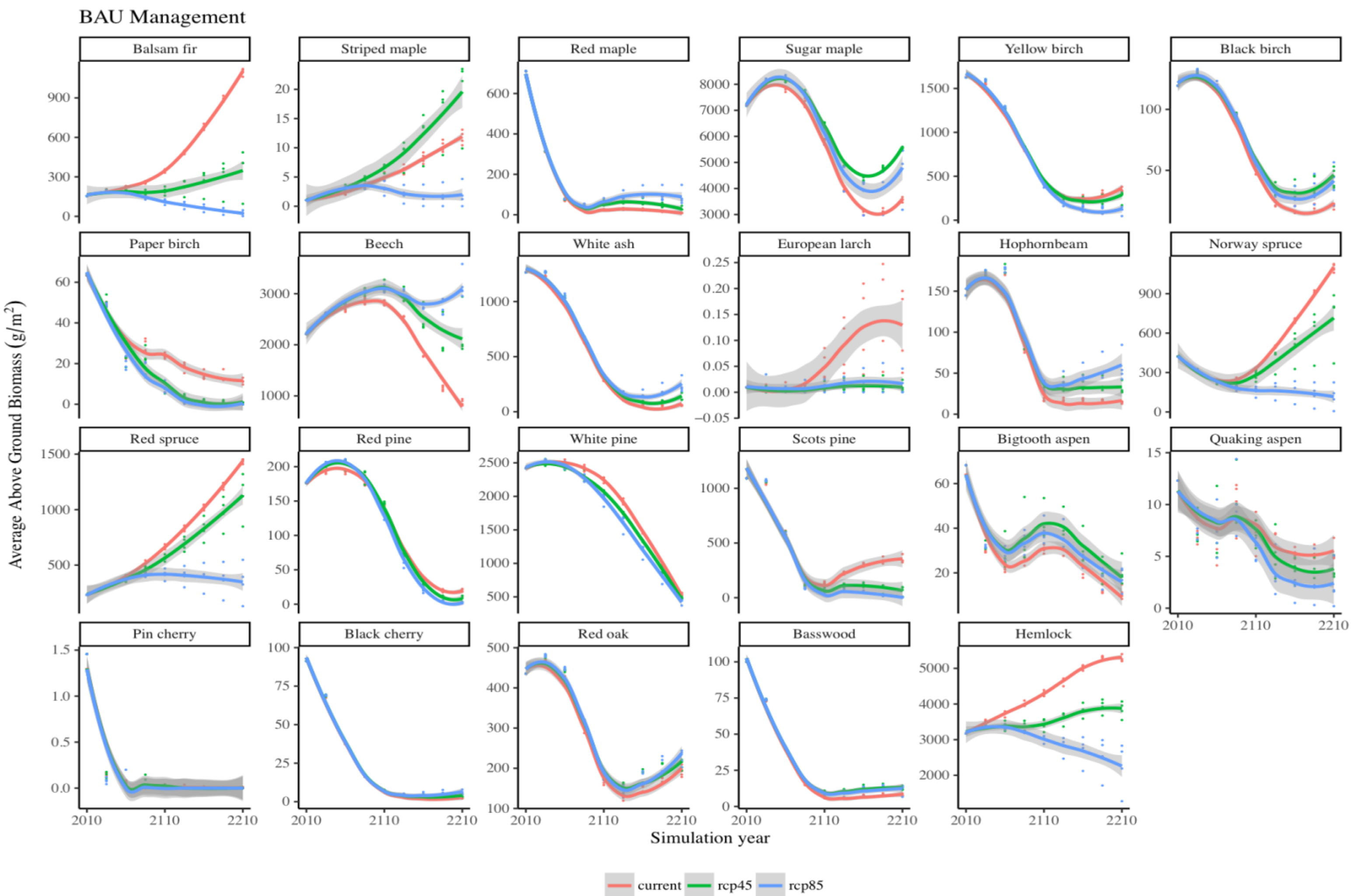


Figure 1: Average total biomass for each tree species simulated under three climate scenarios and BAU management.

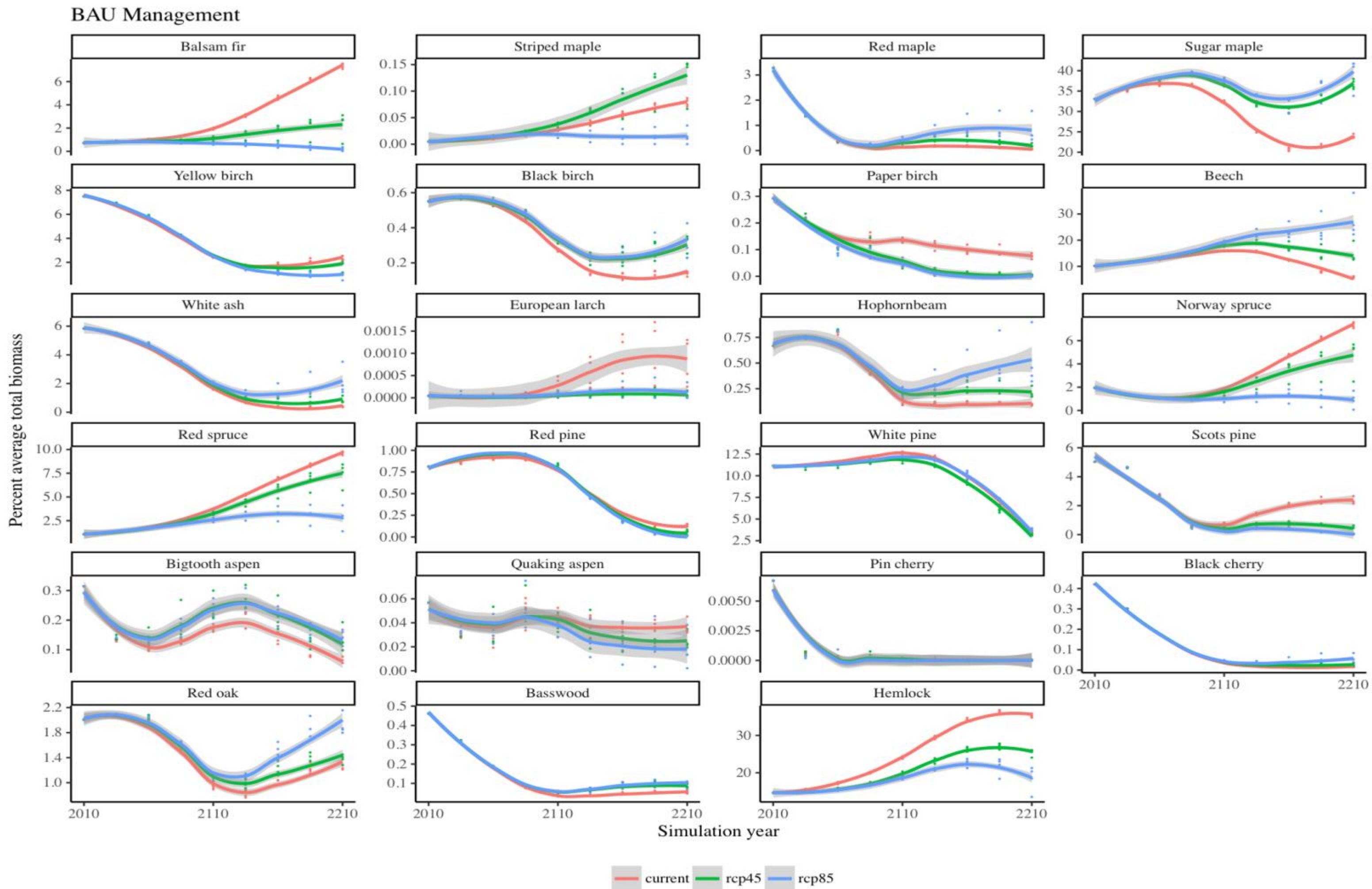


Figure 2: Average percent total biomass for each tree species simulated under three climate scenarios and BAU management

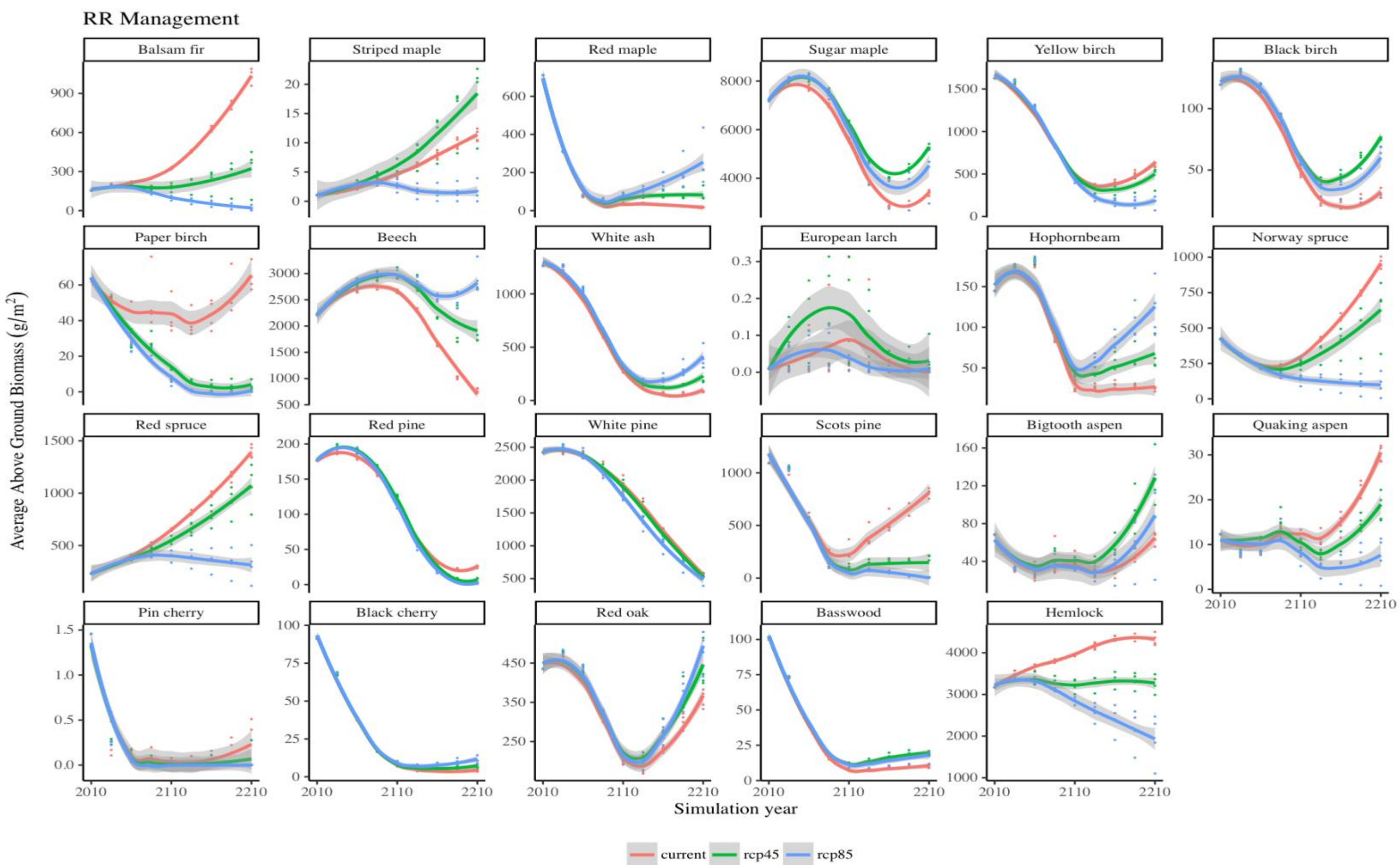


Figure 3: Average total biomass for each tree species simulated under three climate scenarios and RR management.

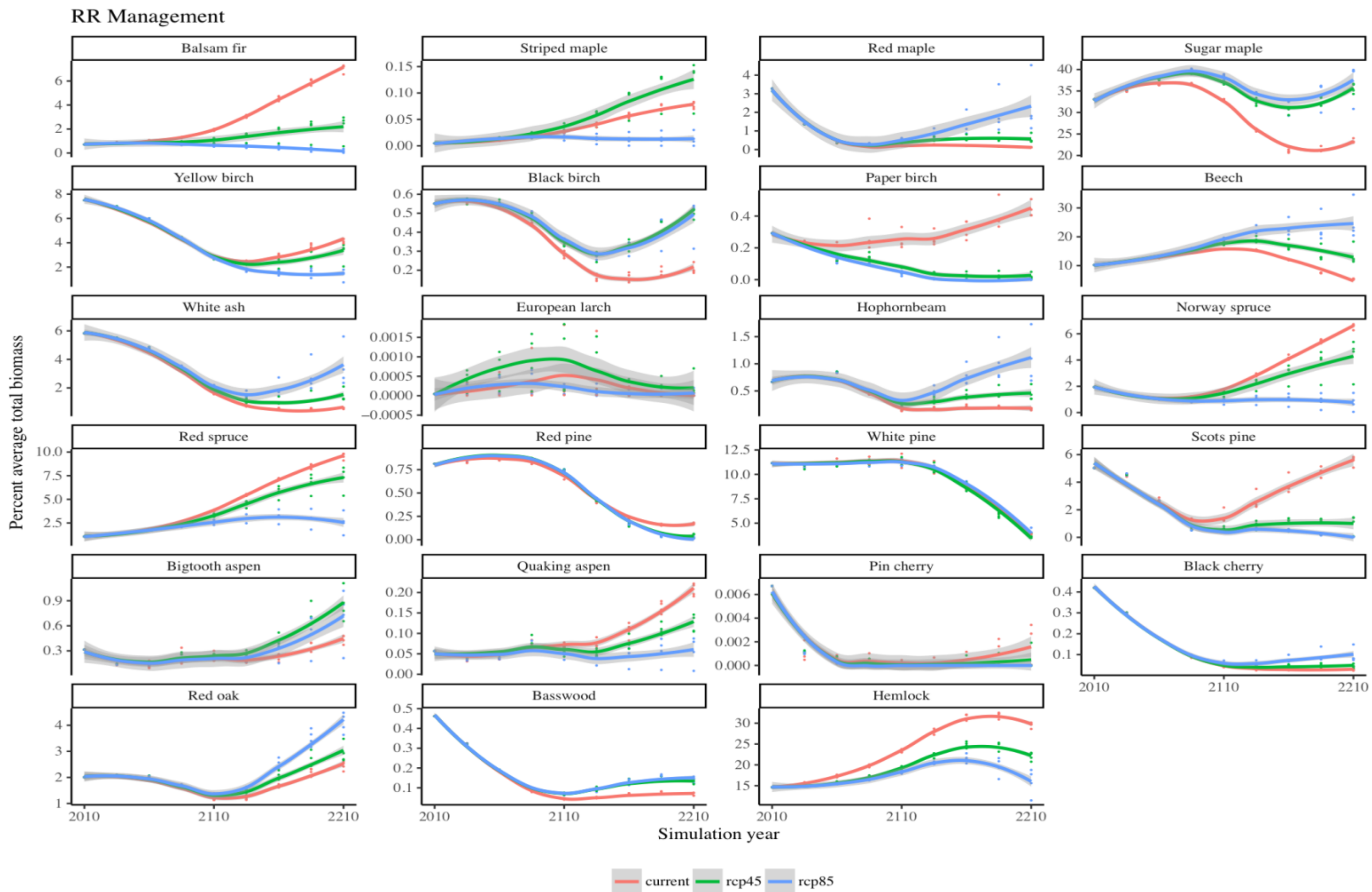


Figure 4: Average percent total biomass for each tree species simulated under three climate scenarios and RR management

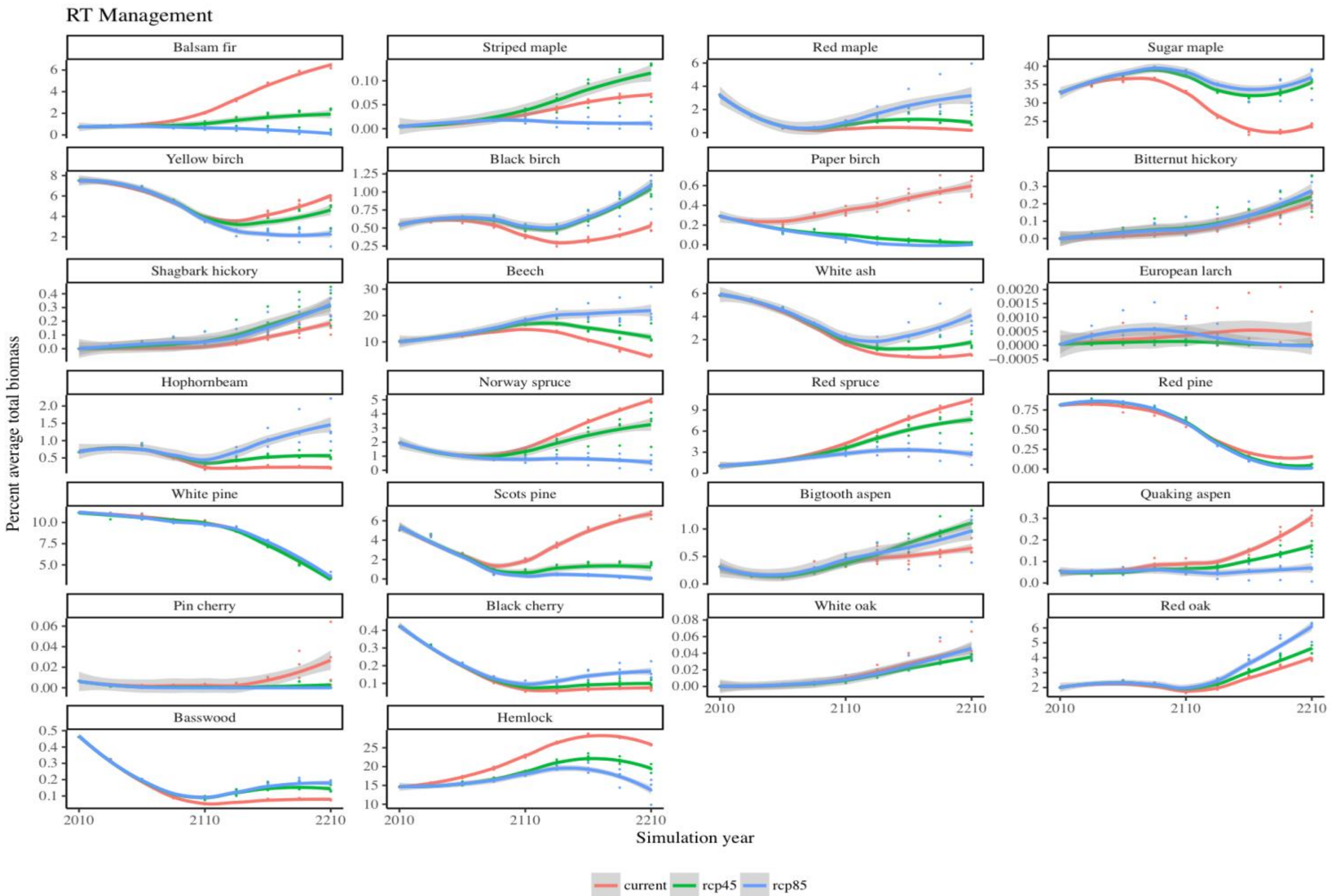


Figure 5: Average total biomass for each tree species simulated under three climate scenarios and RT management.

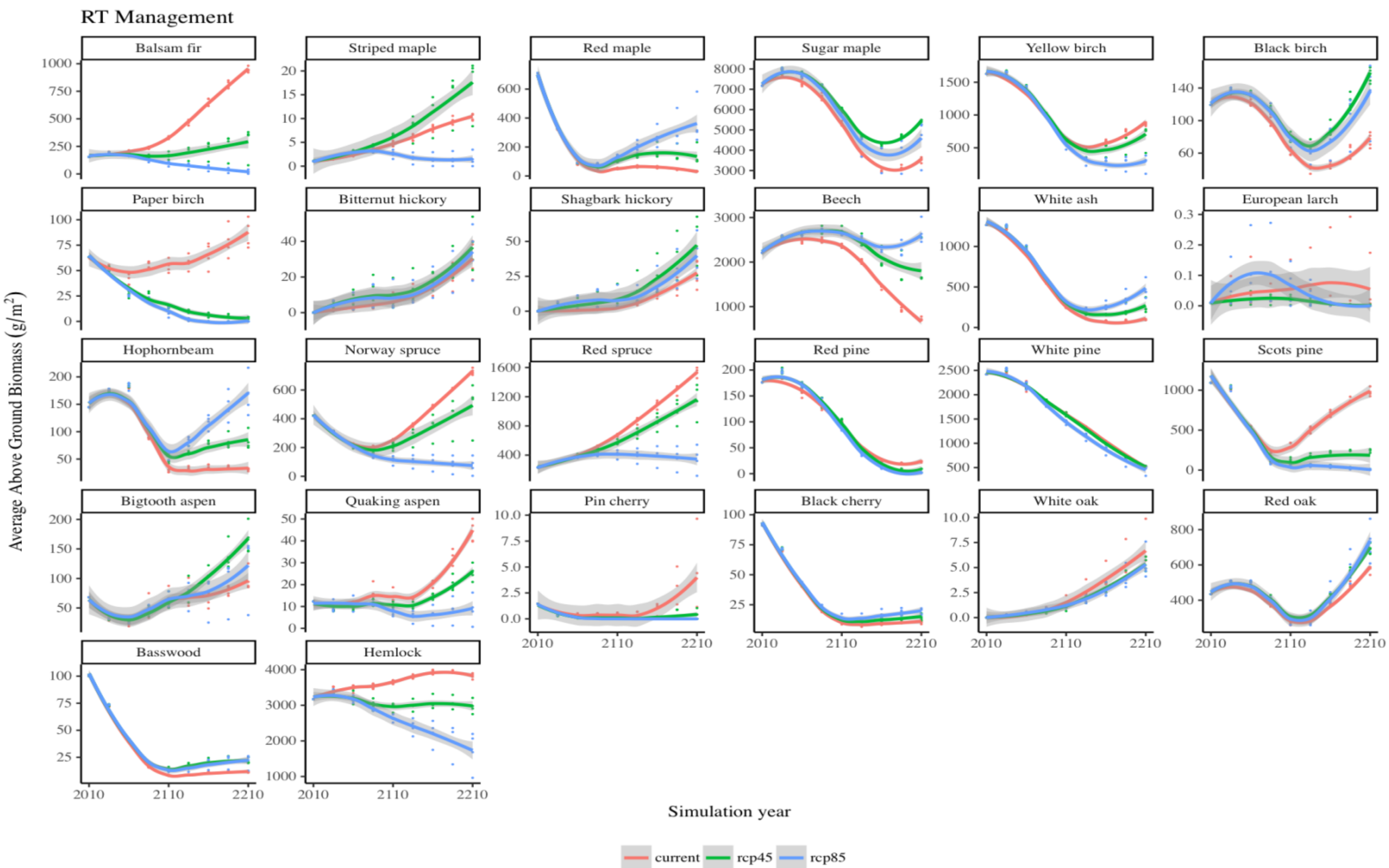


Figure 6: Average percent total biomass for each tree species simulated under three climate scenarios and RT management.

Appendix II: UVA Stand Data: Windsor County, Vermont Management Type Summary

Introduction:

In October 18, 2017 an assessment of Vermont's Use Value Appraisal program was conducted to provide further insights into the current management practices of private forest landowners in Windsor County, Vermont. Below are a series of tables and graphs outlining the findings of an forest management plan data obtained by permission from Vermont's Agency of Natural Resources Department of Forests, Parks, and Recreation.

Stand Type	Sum Acres	% of Total
00 - Not Entered	920.55	0.40%
01 - aspen and/or white birch	1518.42	0.65%
02 - White Pine and/or Red Oak	1297.34	0.56%
03 - white pine	13735.58	5.91%
04 - hemlock	10283.8	4.43%
05 - sugar maple	20050.68	8.63%
06 - beech, birch, sugar maple	78620.24	33.85%
07 - beech, red maple	2934.7	1.26%
08 - spruce	3704.89	1.60%
09 - Spruce and/or Fir	1674.93	0.72%
10 - pioneer species	3990.18	1.72%
11 - Mixed Wood (25%-65% softwood)	77697.93	33.45%
12 - other	14612.18	6.29%
13 - ESTA	953.08	0.41%
14 - Special Places and Sensitive Sites	129.06	0.06%
15 - Miscellaneous	85.4	0.04%
16 - open	38.13	0.02%
17 - Significant Wildlife Habitat	14.1	0.01%
Grand Total	**232261.19**	100.00%

Figure 1: Stand type -Total acres and percent of total

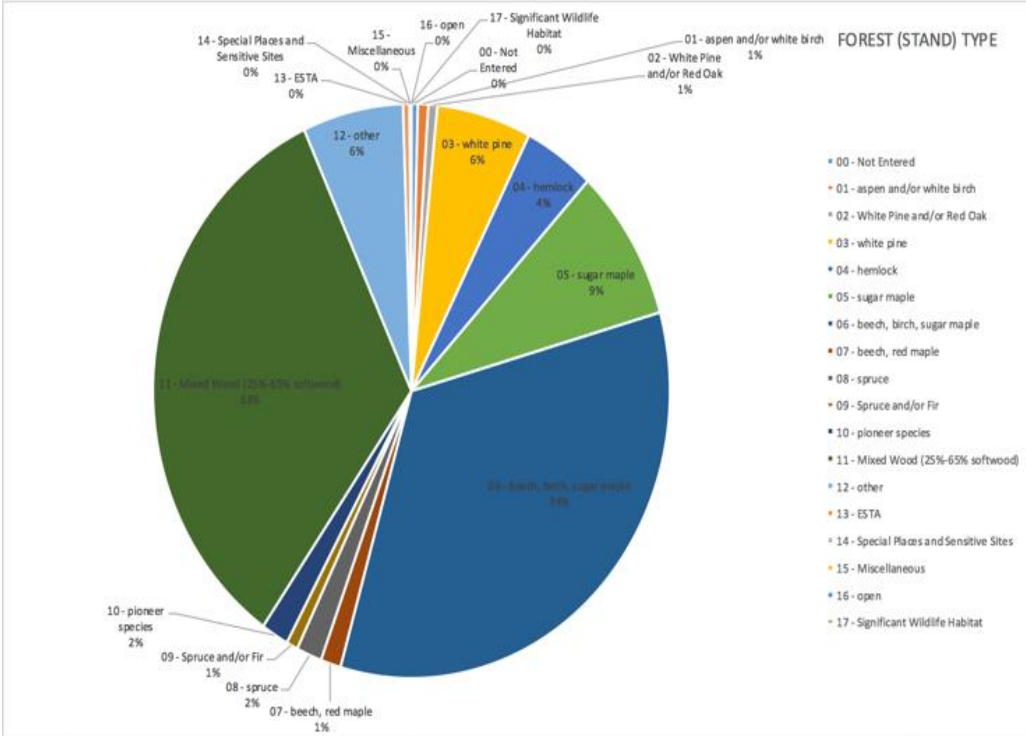


Figure 2: Pie chart of stand type as percentage of total acreage

Table 1: Management Type table - Acres and percent (%) of total

Management Type	ACRES	% of Total
00 - None	8232.88	3.54%
01 - Non-commercial forest stand improvement	3530.7	1.52%
02 - Intermediate thinning	27562.34	11.87%
03 - Shelterwood cut	7784.05	3.35%
04 - Overstory removel cut	4528	1.95%
05 - Clearcut	1280.1	0.55%
06 - Progressive clearcutting	3699.78	1.59%
07 - Single tree selection	65851.36	28.35%
08 - Group selection	19682.07	8.47%
09 - Salvage cut	1972.11	0.85%
10 - Sugarbush management	2893.45	1.25%
11 - Species conversion	164.77	0.07%
12 - No activity	65910.96	28.38%
13 - Other	11271.03	4.85%
14 - Crop tree release	6838.23	2.94%
15 - Invasive species control	1059.36	0.46%
Grand Total	**232261.19**	100.00%

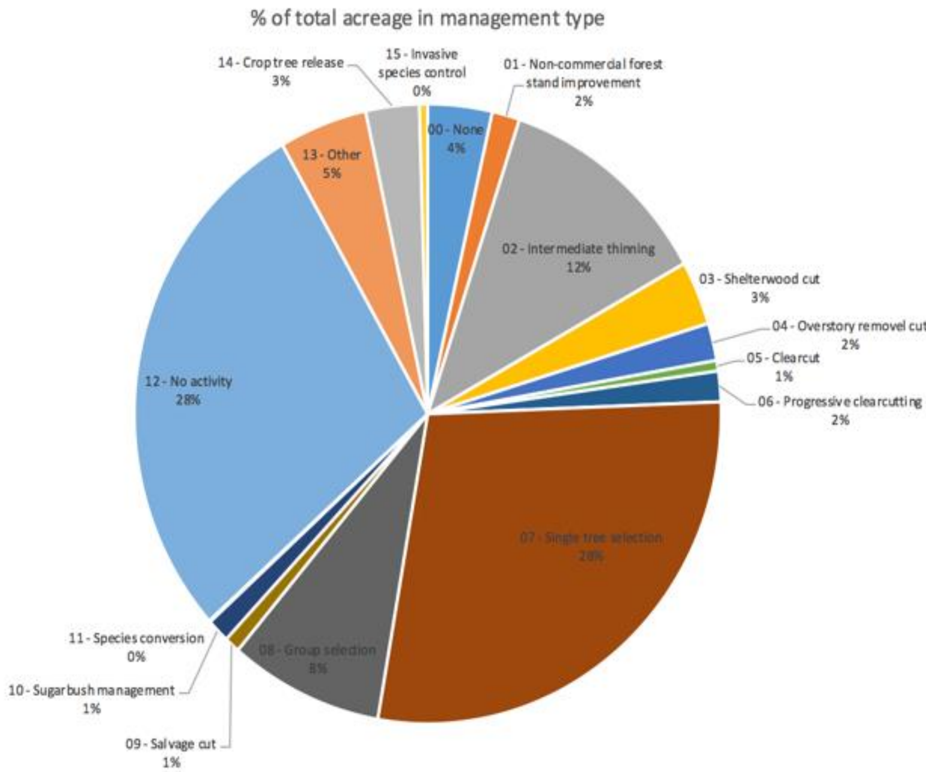


Figure 3: Pie chart of management type as percentage of total acreage

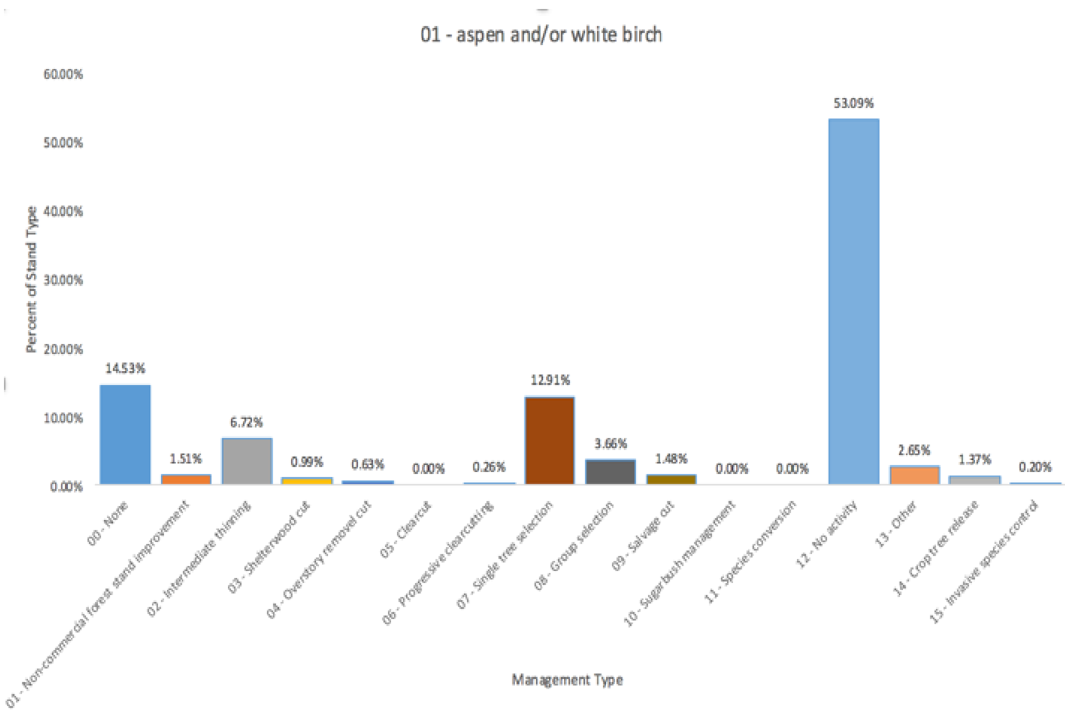


Figure 4: Management within aspen/white birch stands

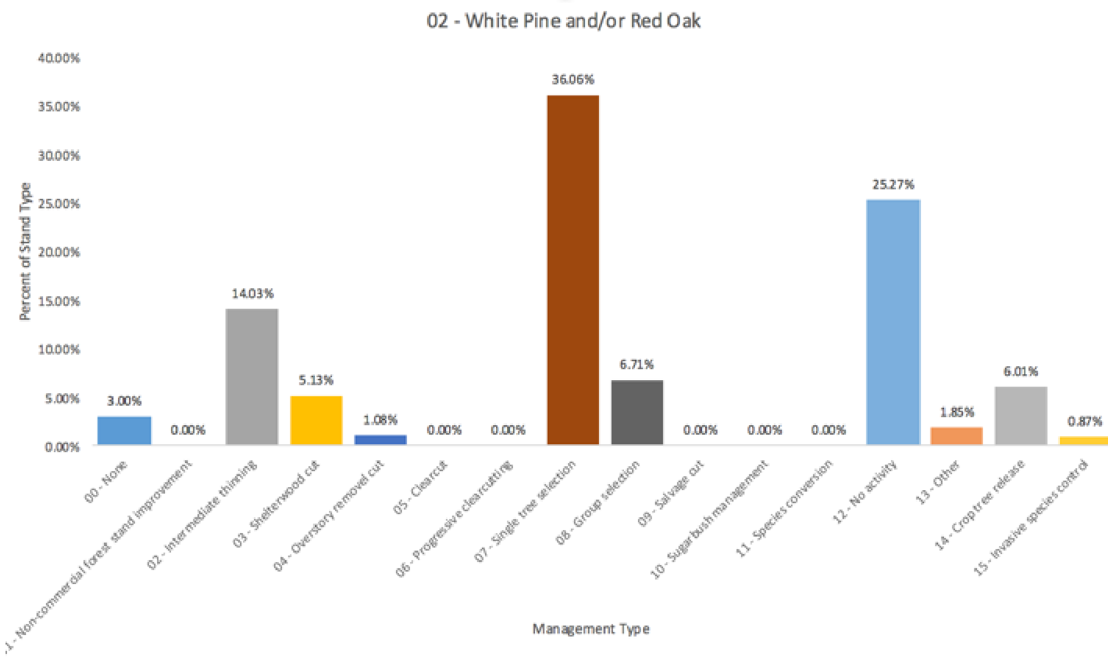


Figure 5: Management within white pine and or red oak stands

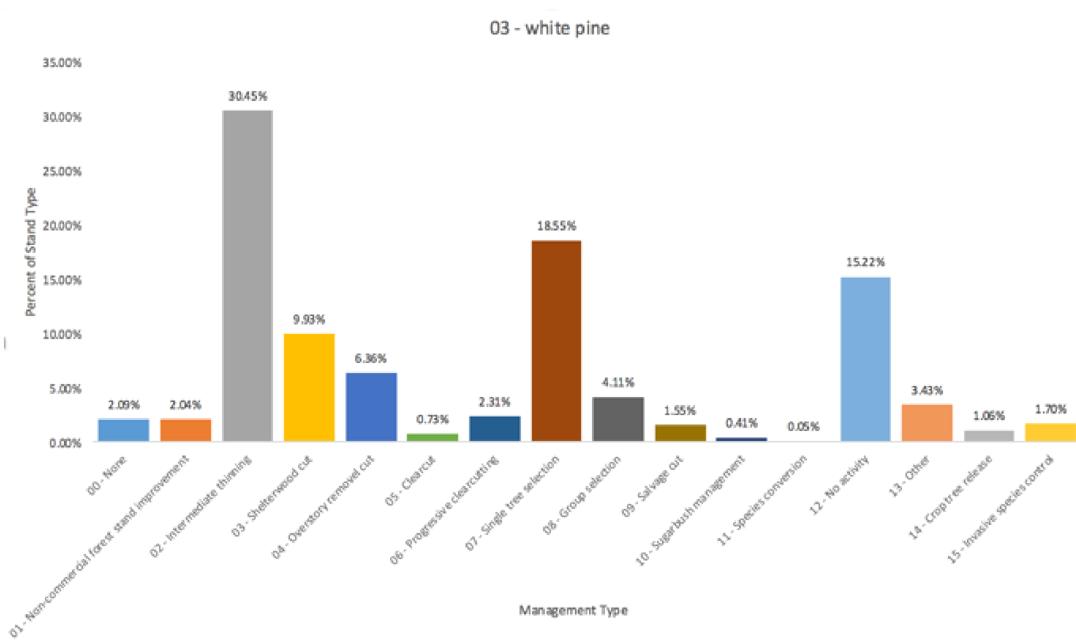


Figure 6: Management within white pine stands

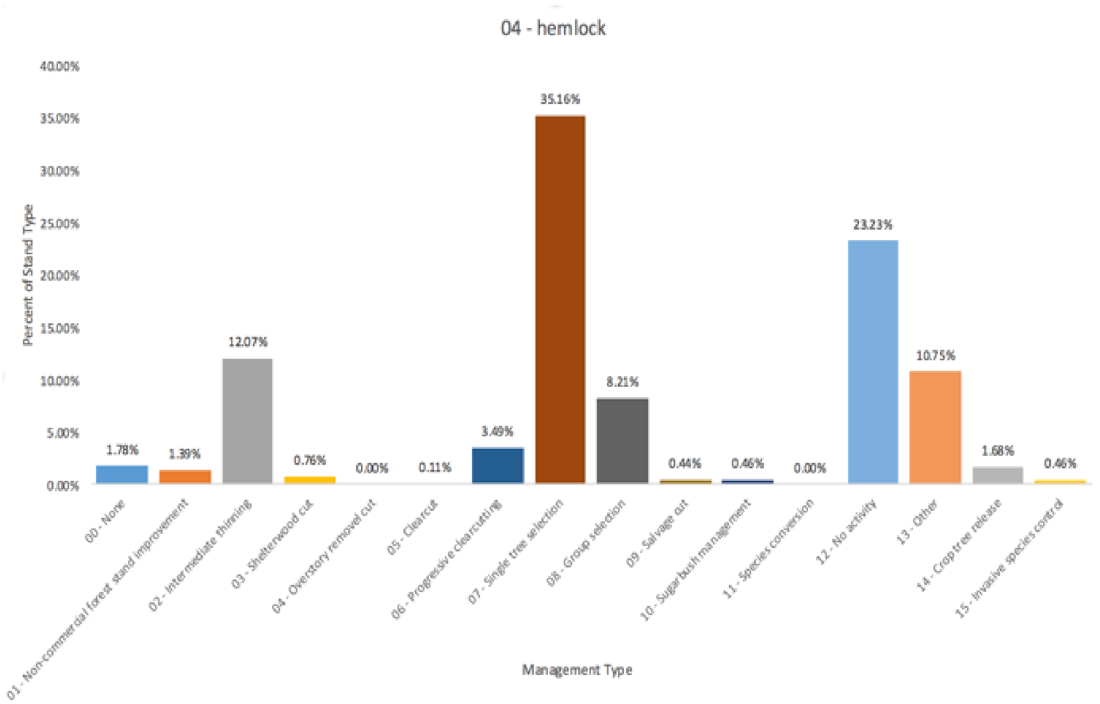


Figure 7: Management within eastern hemlock stands

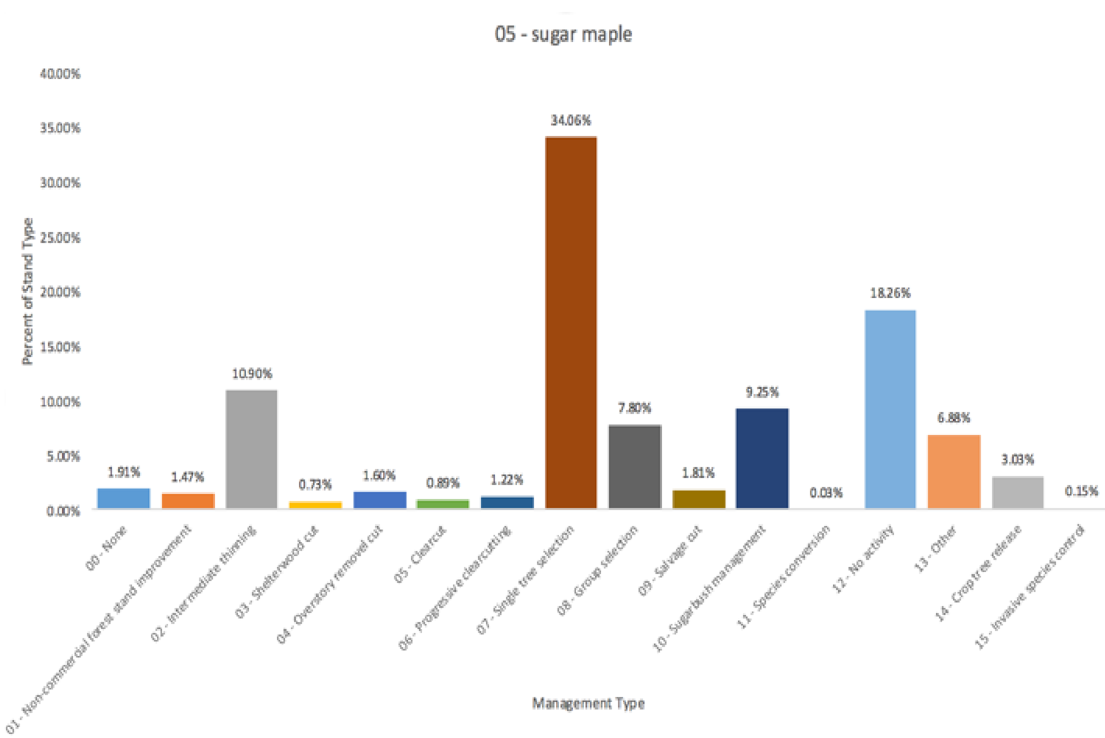


Figure 8: Management within sugar maple stands

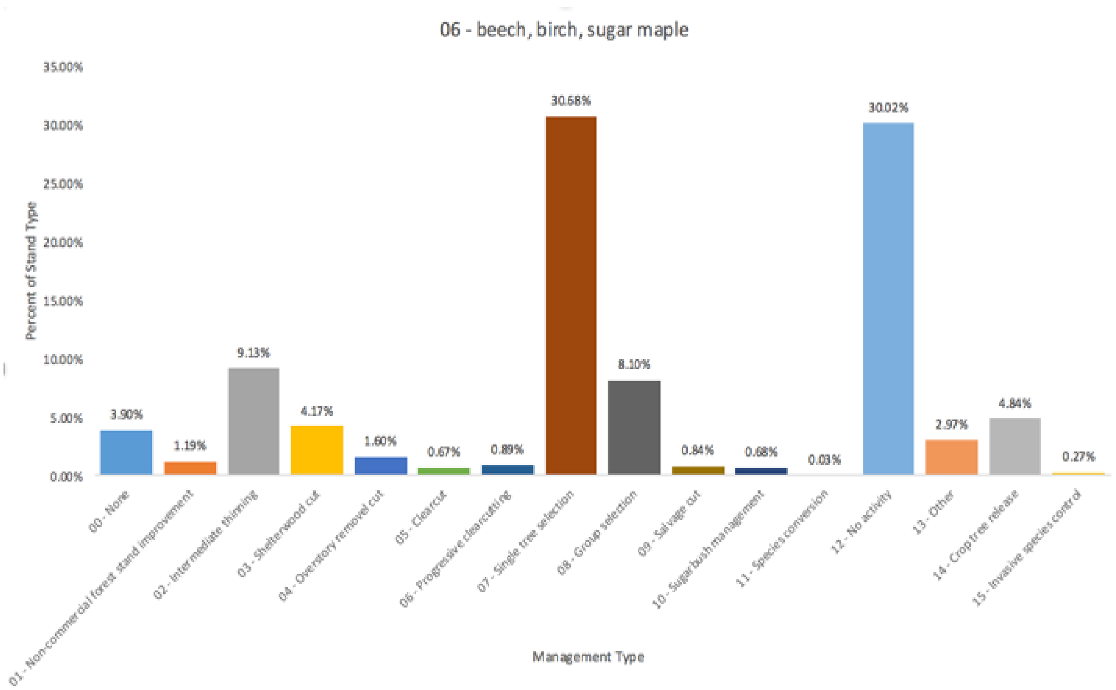


Figure 9: Management within beech, birch, maple stands

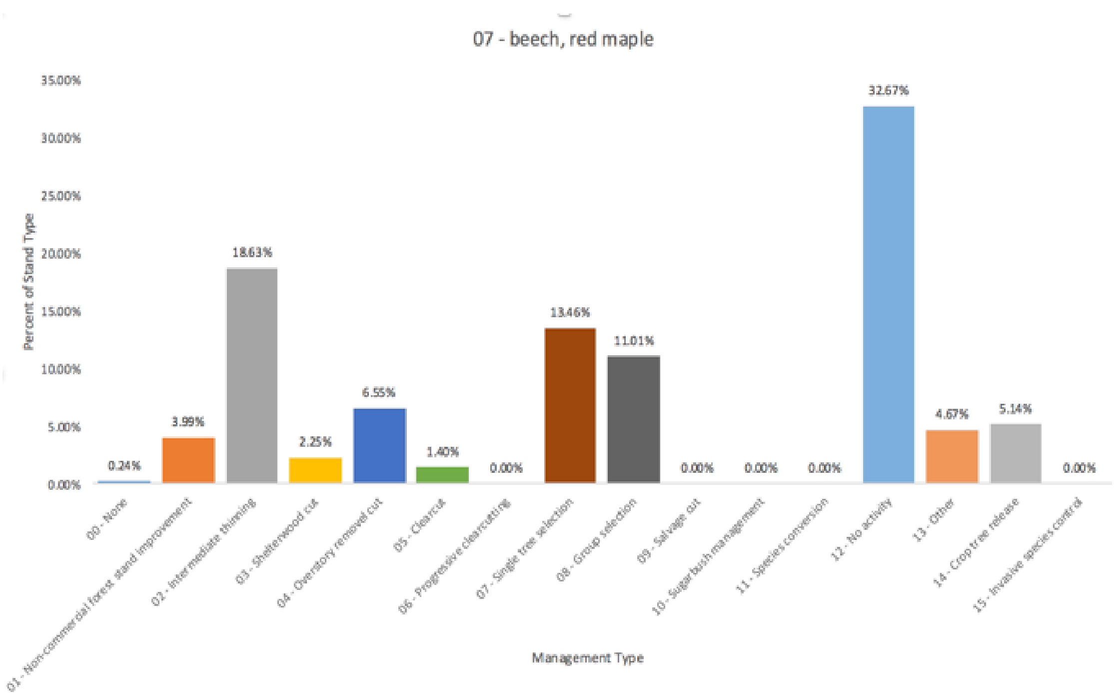


Figure 10: Management within beech, red maple stands

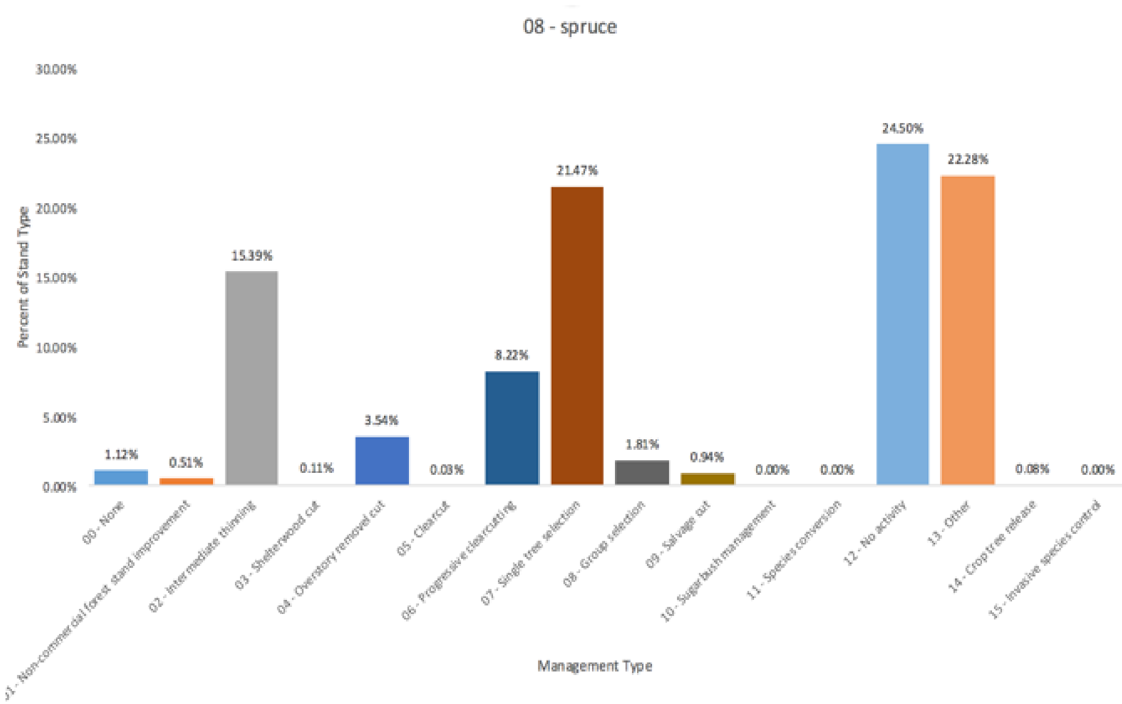


Figure 11: Management within spruce stands

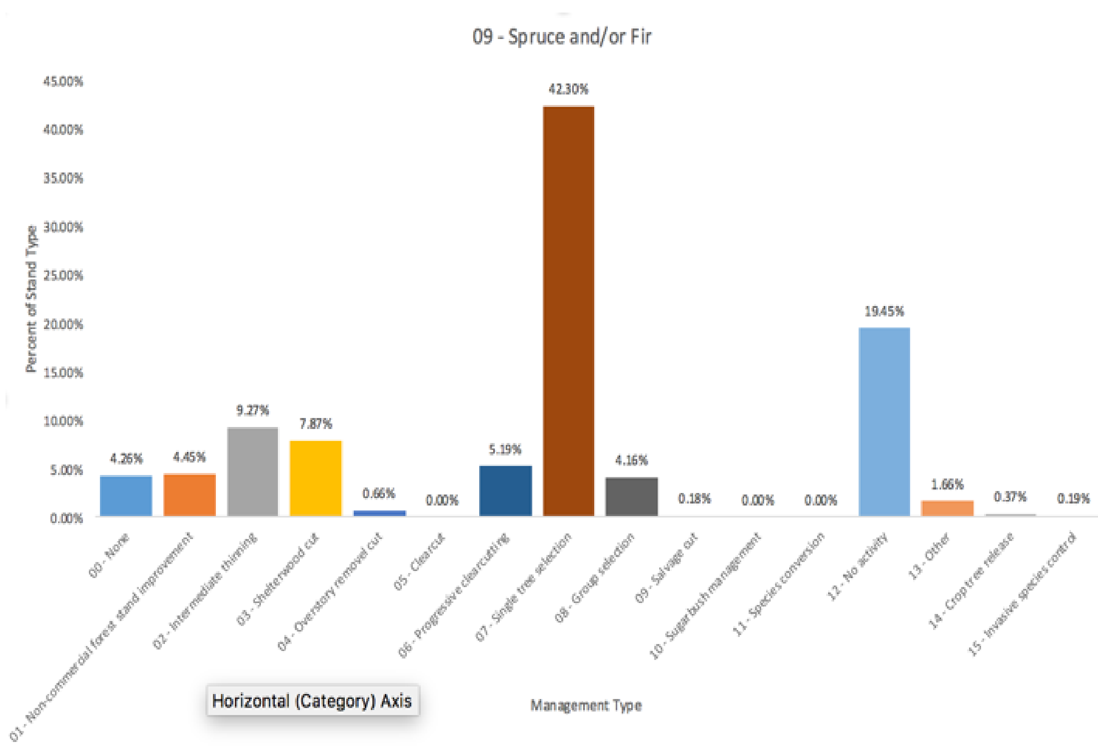


Figure 12: Management in spruce/fir stands

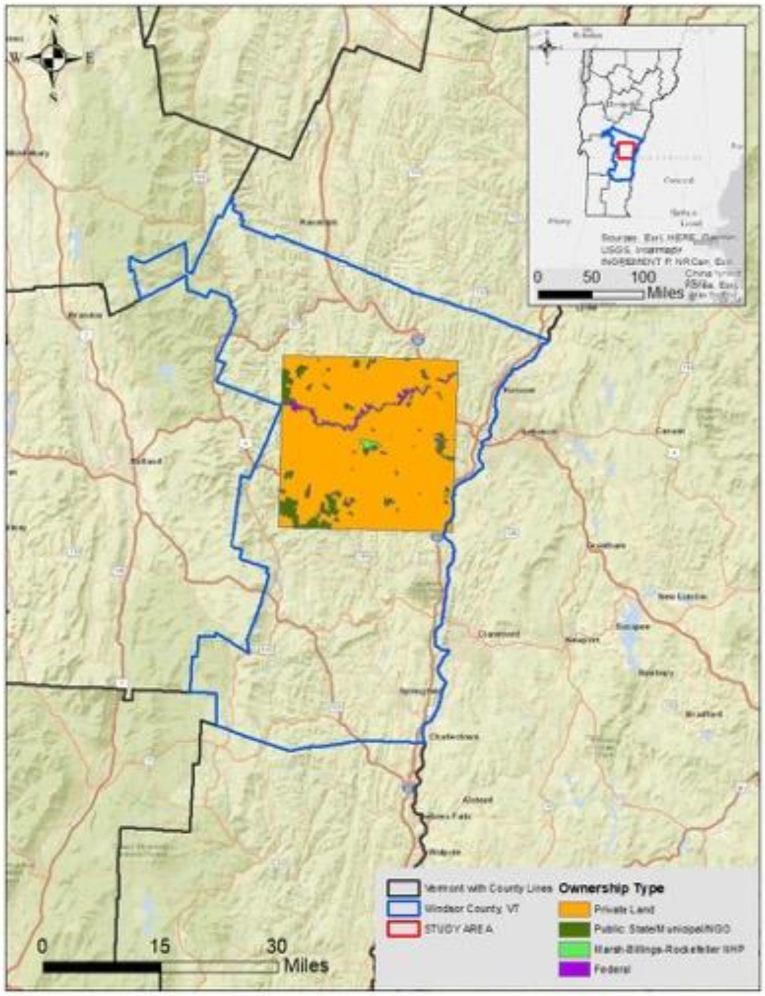
Appendix III: Quantifying Disturbance History: Forest Landscape Simulation Modeling Initialization

Report prepared on: 4/24/18
Prepared by: Matthias Nevins: M.S. Candidate – Forestry, University of Vermont
Landsat Data Processing by: Dr. Jane Foster: Senior Researcher - University of Vermont

Introduction:

Two Landsat derived data packages were used to quantify disturbance patterns within a 58,500 hectare/120,000 acre (+/-) landscape in Windsor County, Vermont (Map 1). These data are the North American Forest Dynamics/NASA Earth Exchange (NAFD-NEX) and the Global Forest Change (v.1.4) 2000-2016 data sets (Goward, 2016; Hansen, 2013). These data sets were used to assess disturbance history within the landscape in order to develop the initial parameters for a forest landscape simulation modeling effort in the same landscape.

The NAFD-NEX data set processes Landsat imagery to classify forest disturbance from year 1986 to year 2010 with a 30m resolution in the conterminous United States. The Global Forest Change data set classifies forest loss from year 2000 to year 2016. These two data sets detect forest harvesting and, to a lesser degree, natural disturbance. The 2005 forest tent caterpillar outbreak which occurred in primarily southern Vermont is the most prominent natural disturbance event detected by these data.



Map 1: Study area located in southeastern Vermont. Landscape located within Windsor County.

Methods:

Each of the data sets were analyzed to quantify the annual disturbance rate and the total area of distinct forest disturbance events (patches) across the landscape. While some forest disturbance events are assumed to be linked to natural disturbances (namely the forest tent caterpillar outbreaks in 2005), the majority of the disturbances measured are related to forest harvesting. By looking at a distribution of patch sizes across the landscape and across ownerships (management areas in LANDIS-II: **See Map 1**), we were able to parameterize typical harvesting intensities within the landscape.

Results:

Based on the NAFD-NEX data set, we observe less than 1% (0.22%) of the landscape being disturbed annually by forest harvesting (Table 1).

Table 1: Annual Disturbance Rates (acres) across ownerships – NAFD-NEX data set

NAFD-NEX Data Set - Disturbance Area (1987-2010) - ACRES				
Ownership	**Total Acres**	**Mean(SE) Acres Disturbed/Year**	**Median**	**% Annually Disturbed**
Public	6623.79	9.11(4.35)	2.67	0.14%
Private	109782.57	245.49(70.69)	152.56	0.22%
Federal	3088.39	4.60(3.52)	0.78	0.15%
Marsh-Billings	518.85	1.83(3.99)	1.00	0.35%
Total Landscape	120013.60	259.92(78.43)	153.79	0.22%

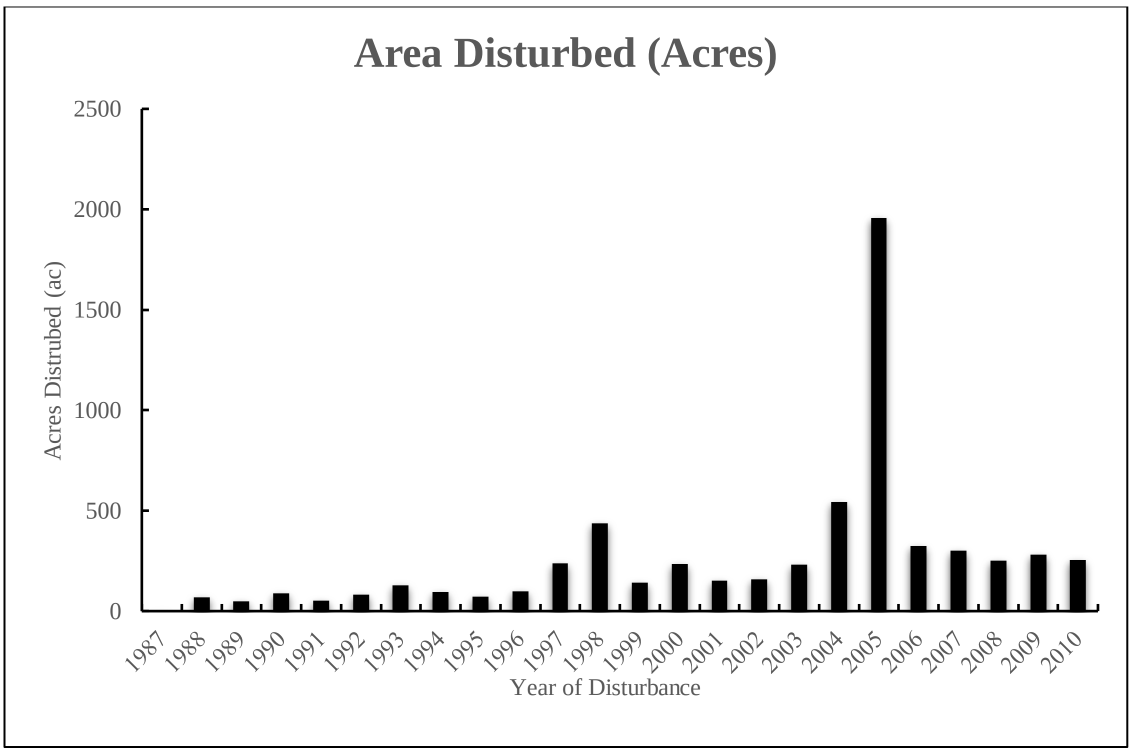


Figure 1: Disturbance rates (acres) for Landscape based on NAFD-NEX data set.

Patch Size Results

The Landsat imagery detects change at the individual pixel (30X30 meter cells) within the landscape. The data was processed to group all pixels associated with a single disturbance event together. This allows us to assess the mean size of individual disturbance events within the landscape. Mean patch size differed across ownerships (Table 2 & Table 3). Mean patch size was largest within public lands (1.13 +/- 0.23 acres) and federal lands had the lowest mean patch sizes (0.40 +/- 0.08 acres) when looking at the Hansen data set. The NAFD-NEX showed larger patch sizes across all ownerships (Table 3). This is due in part to the NAFD-NEX detecting more natural partial disturbances such as the 2005 forest tent caterpillar outbreak.

Table 2: Patch size disturbance patterns for full landscape – Hansen data set

Hansen Data Set - Disturbance Area (2000-2016)					
Ownership	**n (0.22 acre patches)**	**Mean(SE) Size**	**Median**	**Range**	**% Landscape Disturbance**
Public	147	1.13 (0.23)	0.44	0.22 - 26.02	5.1%
Private	2715	0.75 (0.03)	0.44	0.22 - 68.05	94.2%
Federal	11	0.40 (0.08)	0.22	0.22 - 1.11	0.4%
Marsh-Billings	9	0.57 (0.13)	0.44	0.22 - 1.33	0.3%
Total Landscape	2882	0.76(0.03)	0.44	0.22 - 68.05	100.0%

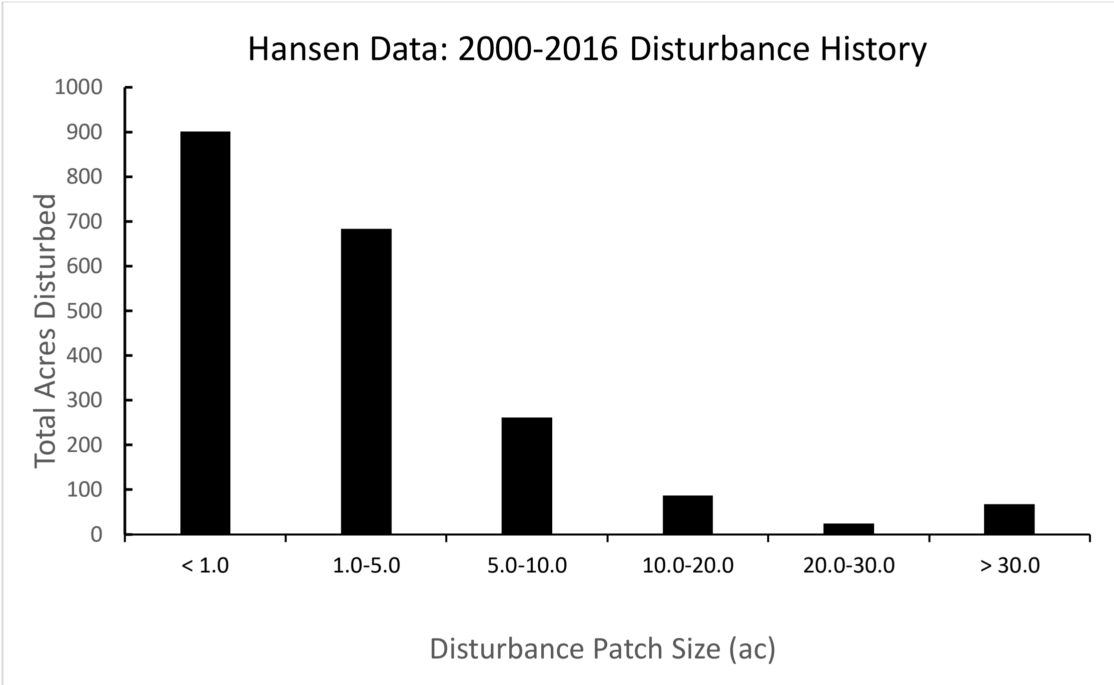


Figure 2: Patch size disturbance (acres) full landscape – Hansen data set

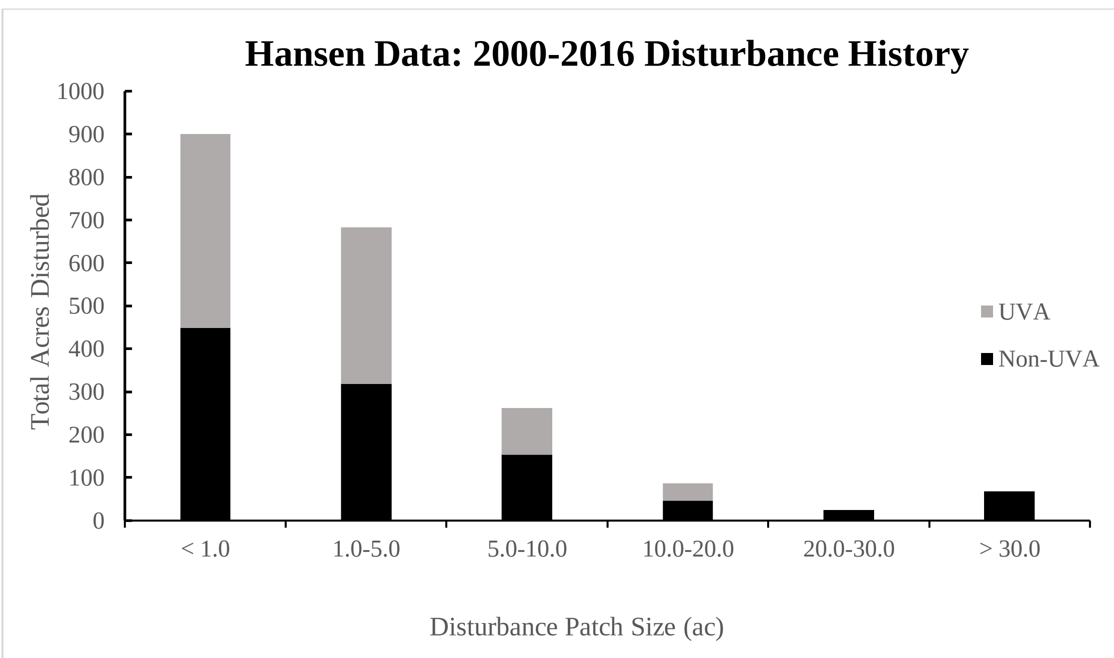


Figure 3: Patch size disturbance (acres) private lands – Hansen data set. Use value appraisal lands shown in comparison to non-enrolled private lands.

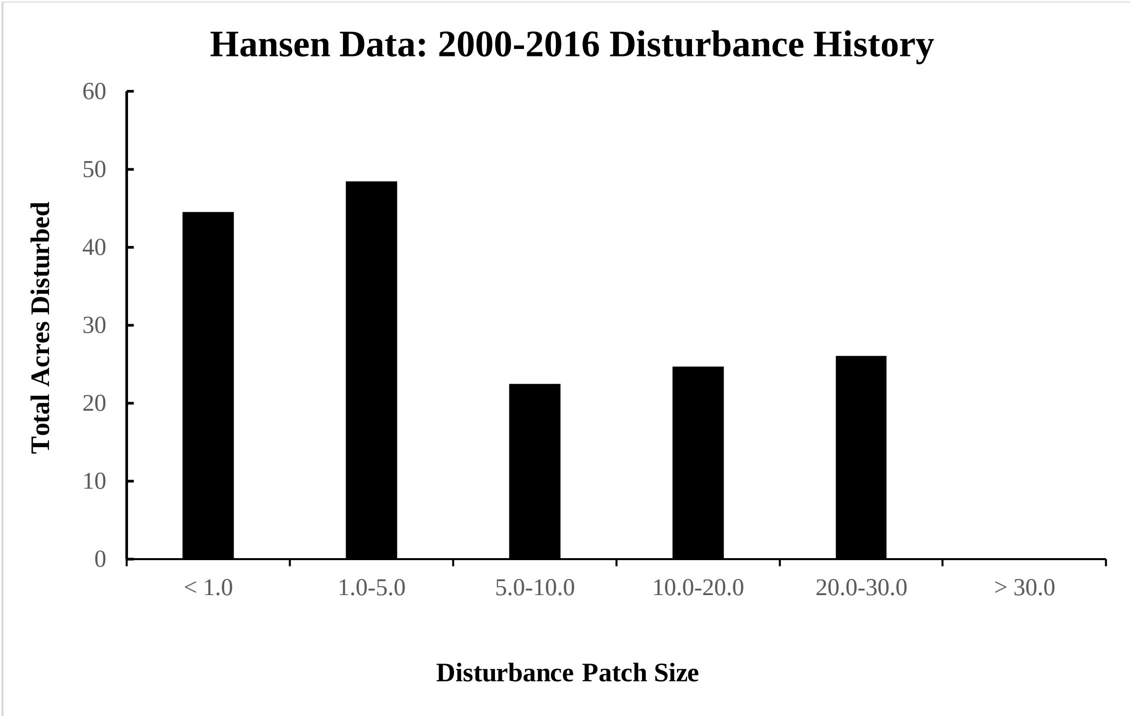


Figure 4: Patch size disturbance (acres) PUBLIC LANDS – Hansen data set.

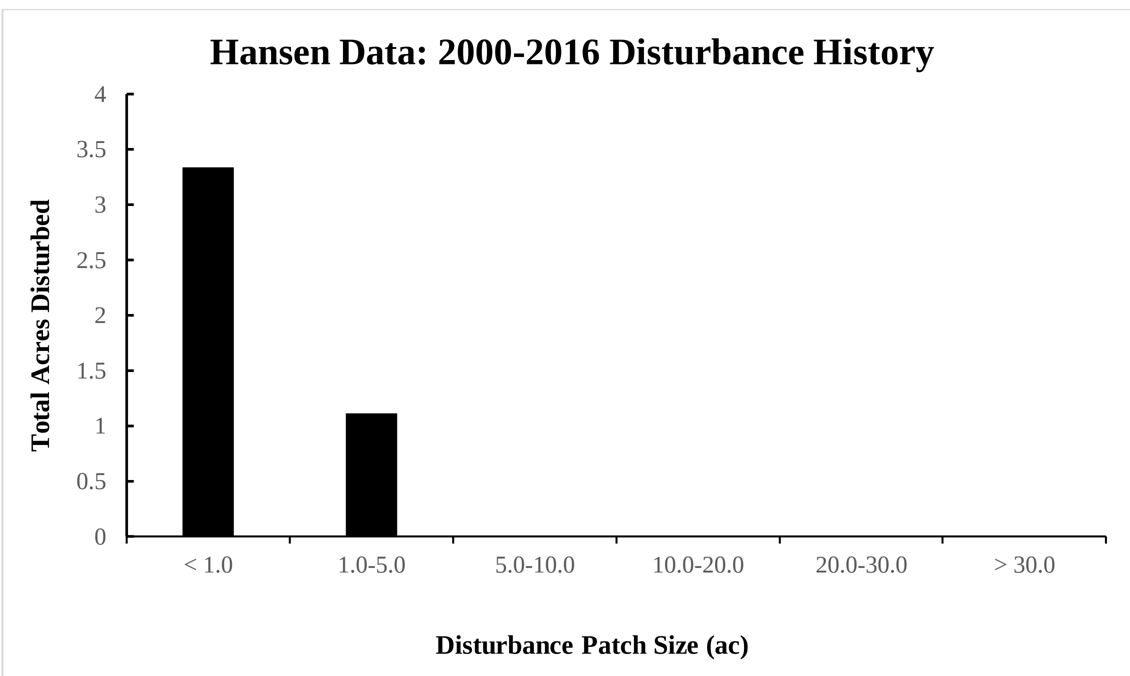


Figure 5: Patch size disturbance (acres) FEDERAL LANDS: Appalachian Trail Corridor – Hansen data set.

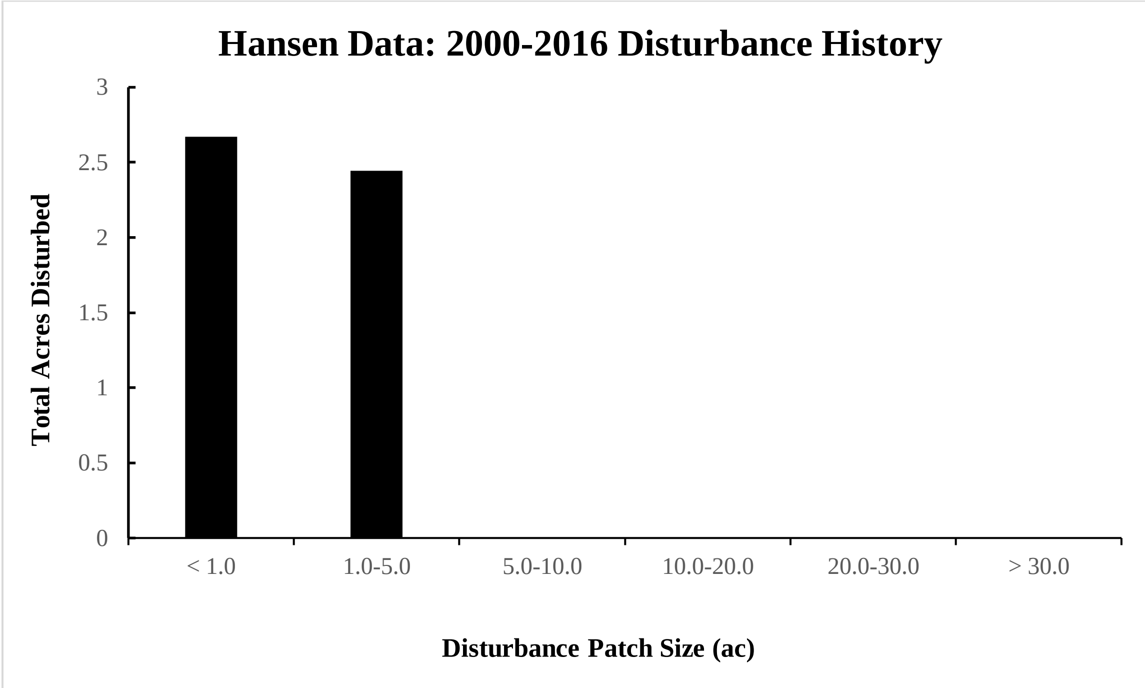


Figure 6: Patch size disturbance (acres) MARSH-BILLINGS-ROCKEFELLER NATIONAL HISTORICAL PARK – Hansen data set.

Table 3: Patch size disturbance (acres) full landscape – NAFD-NEX data set

NAFD-NEX Disturbance Area (1987-2010) -ACRES					
Ownership	**n (0.22 acre patches)**	**Mean(SE) size**	**Median**	**Range**	**% Landscape Disturbance**
Public	165	1.32(0.25)	0.44	0.22 - 28.46	3.2%
Private	4842	1.17(0.04)	0.44	0.22 - 45.38	94.8%
Federal	86	1.28(0.44)	0.44	0.22 - 36.91	1.7%
Marsh-Billings	15	2.97(2.18)	0.44	0.22 - 33.14	0.3%
Total Landscape	5108	1.18(0.06)	0.44	0.22 - 80.06	100.0%

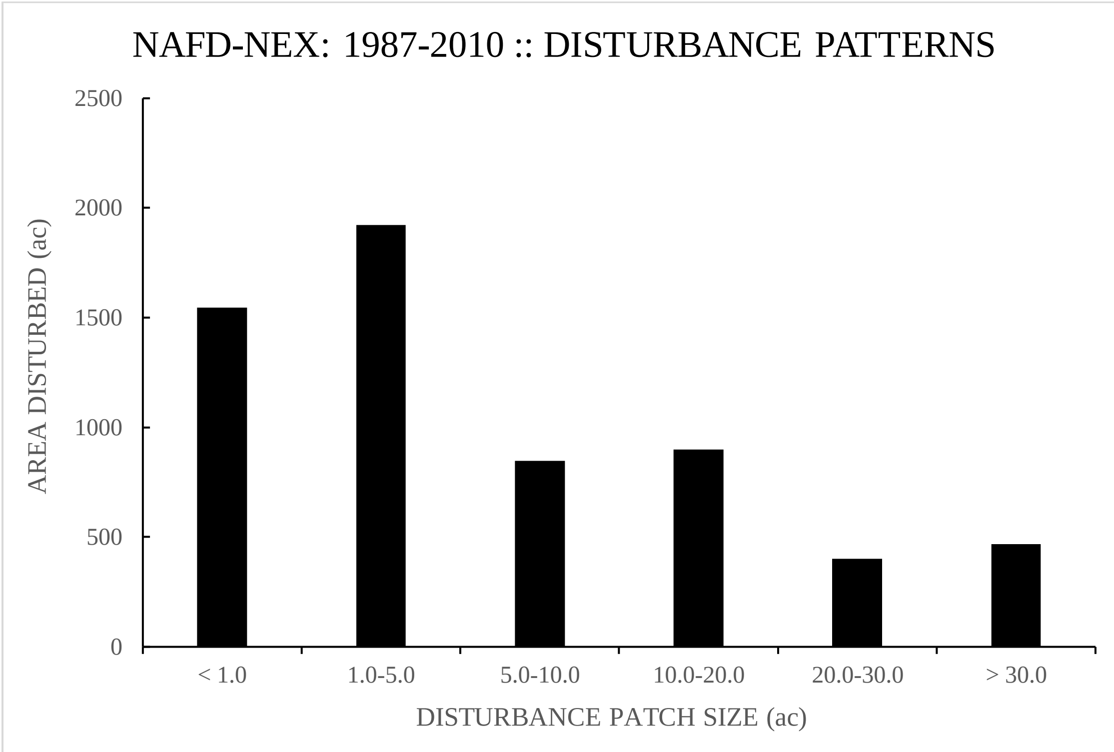


Figure 7: Patch size disturbance (acres) full landscape – NAFD-NEX data set

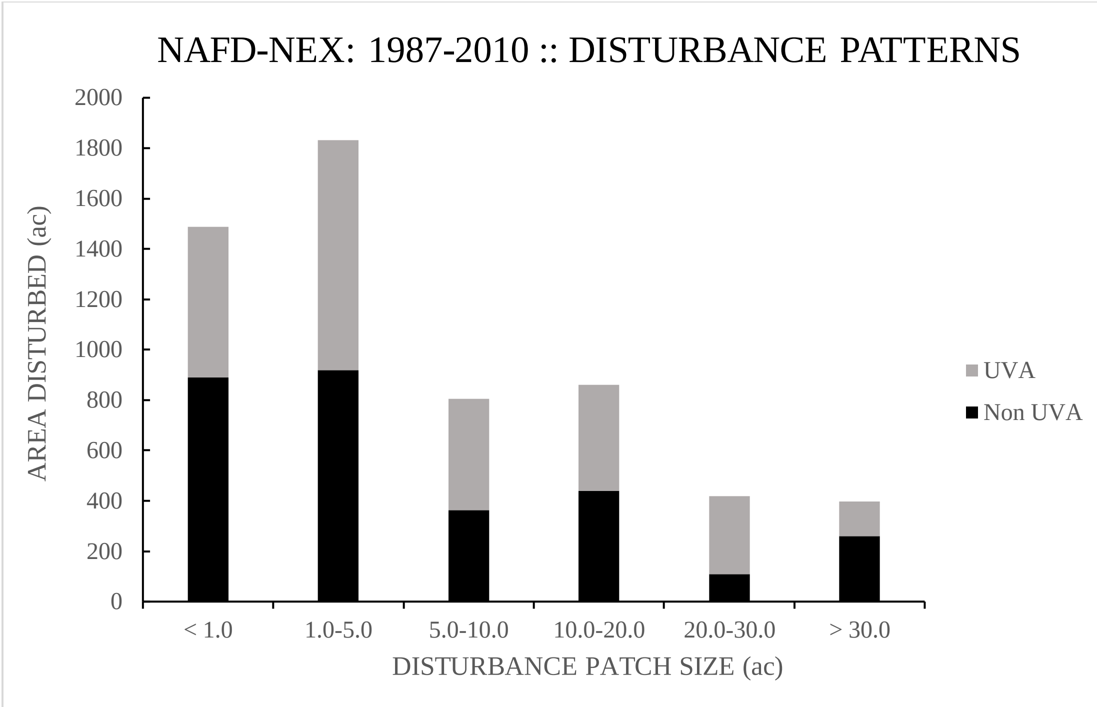


Figure 8: Patch size disturbance (acres) PRIVATE LANDS – NAFD-NEX data set. Use value appraisal lands shown in comparison to non-enrolled private lands.

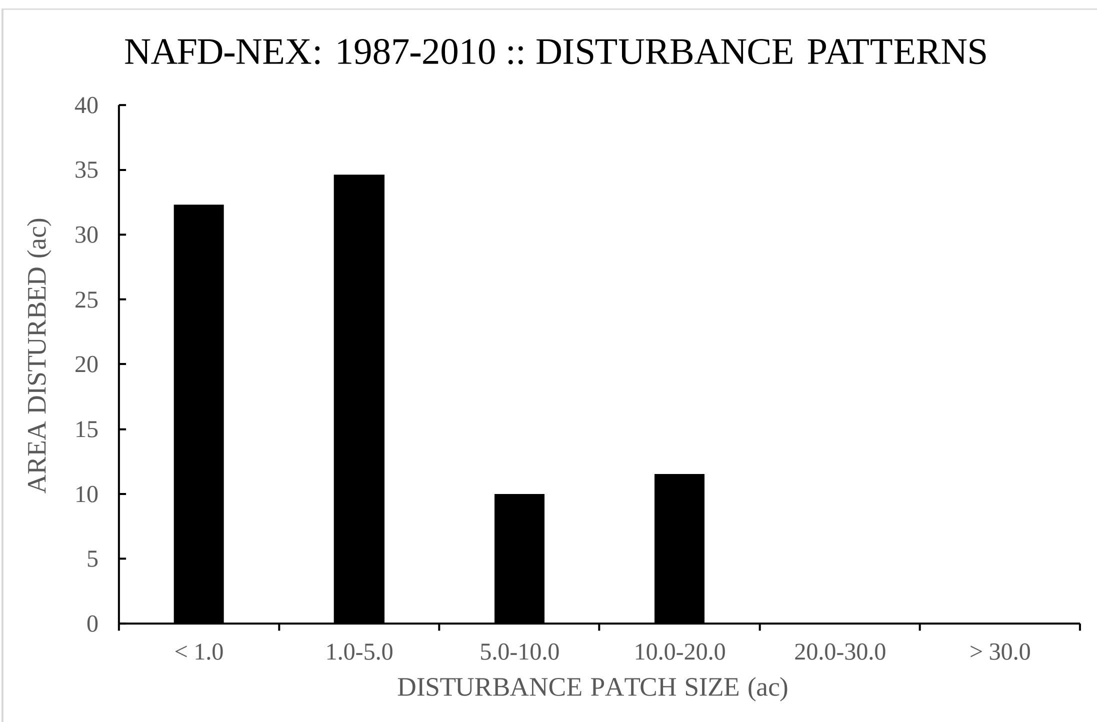


Figure 9: Patch size disturbance (acres) PUBLIC LANDS – NAFD-NEX data set

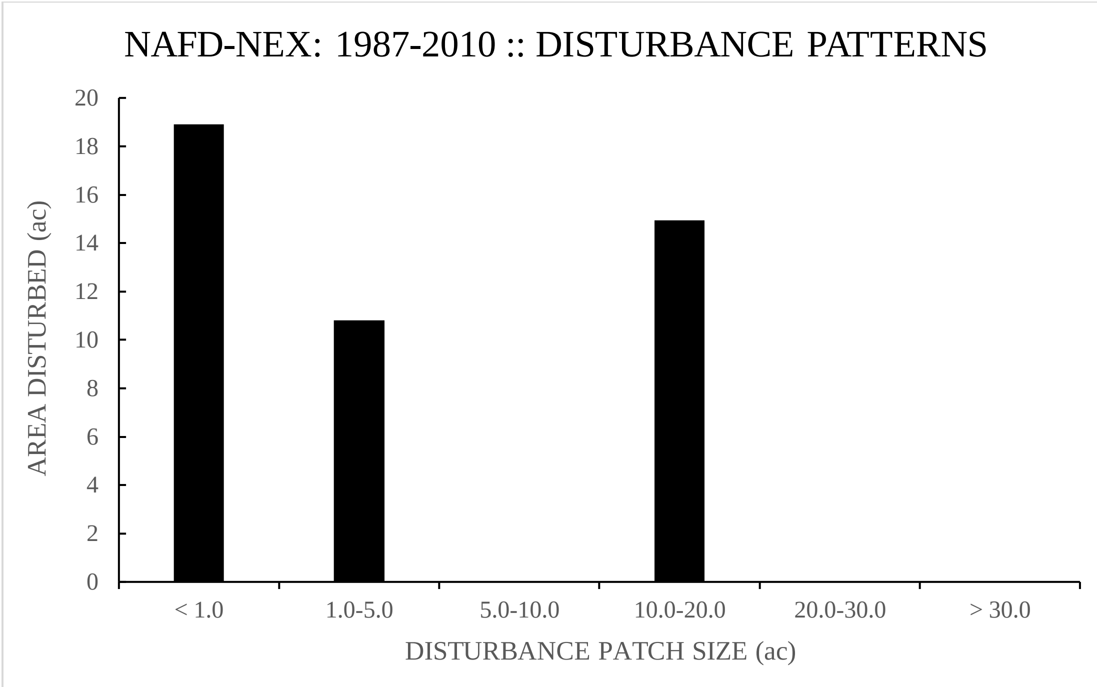


Figure 10: Patch size disturbance (acres) FEDERAL LANDS: Appalachian Trail Corridor – NAFD-NEX data set

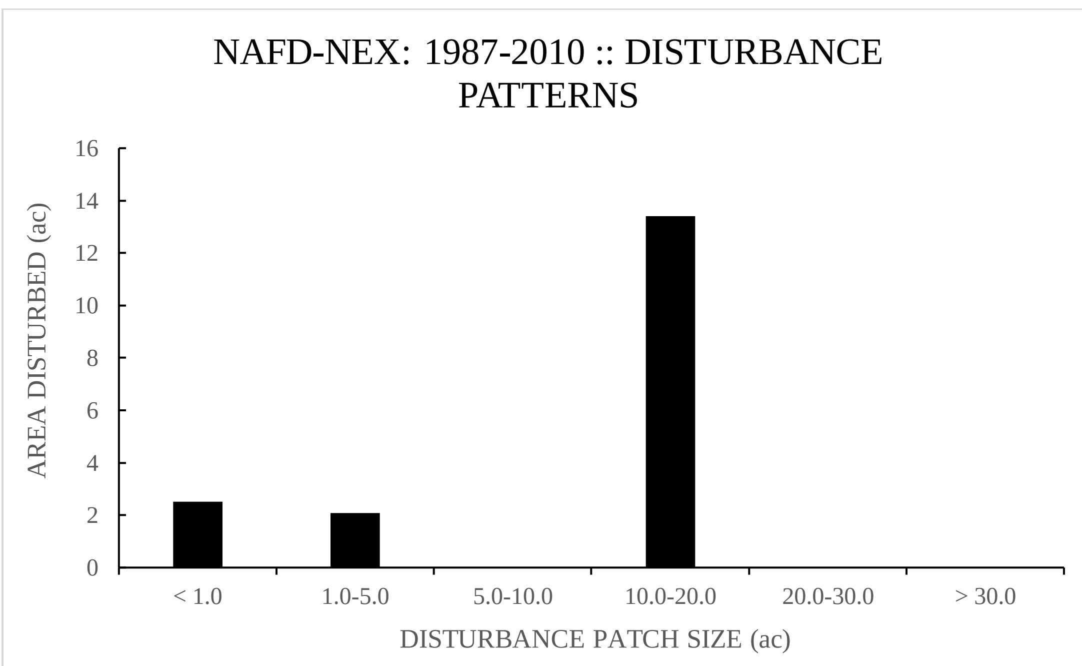


Figure 11: Patch size disturbance (acres) MARSH-BILLINGS-ROCKEFELLER NATIONAL HISTORICAL PARK – NAFD-NEX data set

Discussion

The distribution of patch sizes across ownerships in both data sets show that while the typical range of patch sizes vary from 0.22 acres to 68.05 acres, average patch sizes across ownerships are less than 1-acre in the Hansen data set and less than 2-acres in NAFD-NEX data set. These findings reflect the understanding that typical annual disturbance rates for this region are minor. Forest management within the landscape typically utilizes partial cutting (selection methods) and intermediate treatments (thinning). Patch cutting or clearcutting while present on the landscape are not common. The distributions of patch sizes presented here show that forest management within the landscape is characterized by small scale disturbance. These findings well help inform the simulation of forest management and its influence on long-term forest growth and composition.

Appendix IV: Windsor County Vermont Use Value Appraisal FMAR Analysis

Prepared by:
Matthias Nevins
M.S. Candidate – Forestry, University of Vermont

Introduction

Annual harvest rates were analyzed for Windsor County, Vermont using Forest Management Activities Report (FMAR) data. These data are collected by the State of Vermont for all private properties enrolled in the current use value appraisal (UVA) program. Landowners who are enrolled in this program work with state and private consulting foresters to develop long-term forest management objectives for their property and agree to restrict all residential development for a ten year period in exchange for reduced or preferential property tax treatment on the enrolled acreage. In order to ensure the long-term management of the enrolled properties, landowners are required to conduct forest management activities within the ten-year enrollment period. These management activities are reported to state and include estimates of harvested timber volume.

A subset of these data from Windsor County, Vermont was used to determine the average timber volume removed per acre over a five-year period. This analysis will be used to inform forest landscape simulation modeling efforts looking to evaluate the potential long term influence of disturbance and forest management of forest development.

Methods

FMAR data for Windsor County, Vermont were analyzed to determine the average timber volume removed per acre for enrolled parcels over a five-year period. The years 2012 to 2016 were used because the data was most complete for these years.

Harvest volumes for each product class were converted to cubic feet (ft^3) and cords in order to assess the total timber volume removed.

The total harvested volume in each year was calculated for each enrolled property. This volume was then divided by the total acreage of the property. We were unable to compare the reported harvest volumes to the associated stand acreage in this analysis. We present a distribution of harvest volume/acre as a means of representing typical silvicultural treatments. These comparisons are made in Table 2 in the discussion and will further assist the development of the simulation models.

Results

The annual harvest rate from 2012-2016 in Windsor County, Vermont is 125.782 ft^3/acre (+/-9.07) and 7.55 cords/acre (+/-0.54).

Annual Harvest Rate (Five Year Mean)		
Statistic	**Ft^3/acre**	**Cords/Acre**
Mean	125.782	7.550
SD	20.280	1.217
SE	9.070	0.544
Min	0.047	0.003
Max	4711.450	282.800
Median	35.090	2.110

Table 1: Annual Harvest Rates (Five year mean) Windsor County based on UVA FMAR

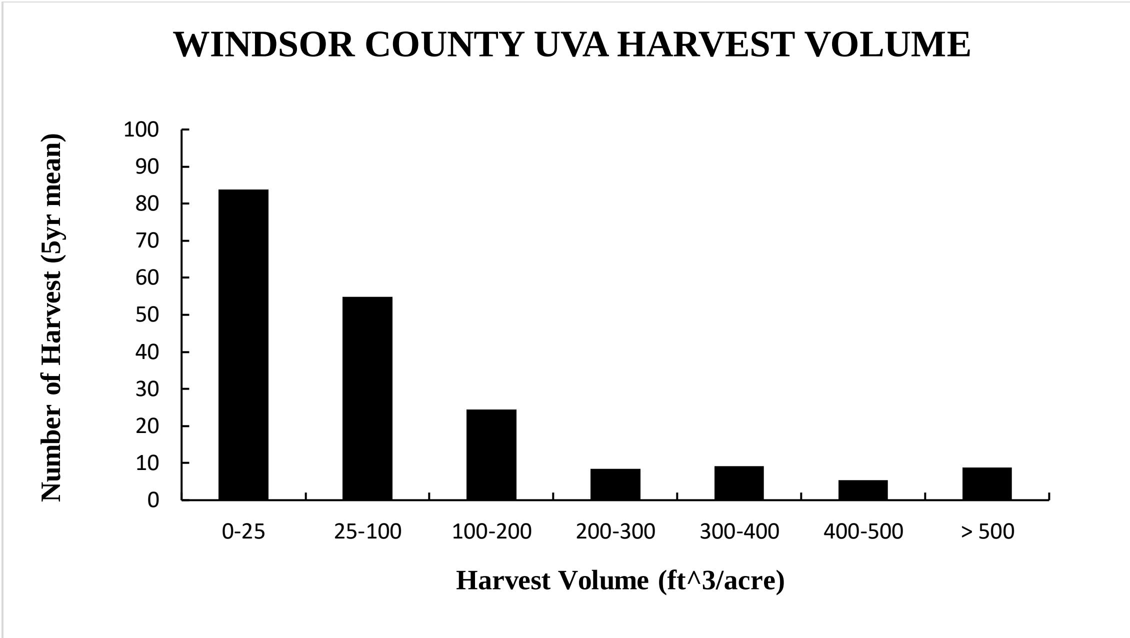


Figure 1: Distribution of harvest volumes (ft^3/acre) for Windsor county based on a five year mean 2012-2016.

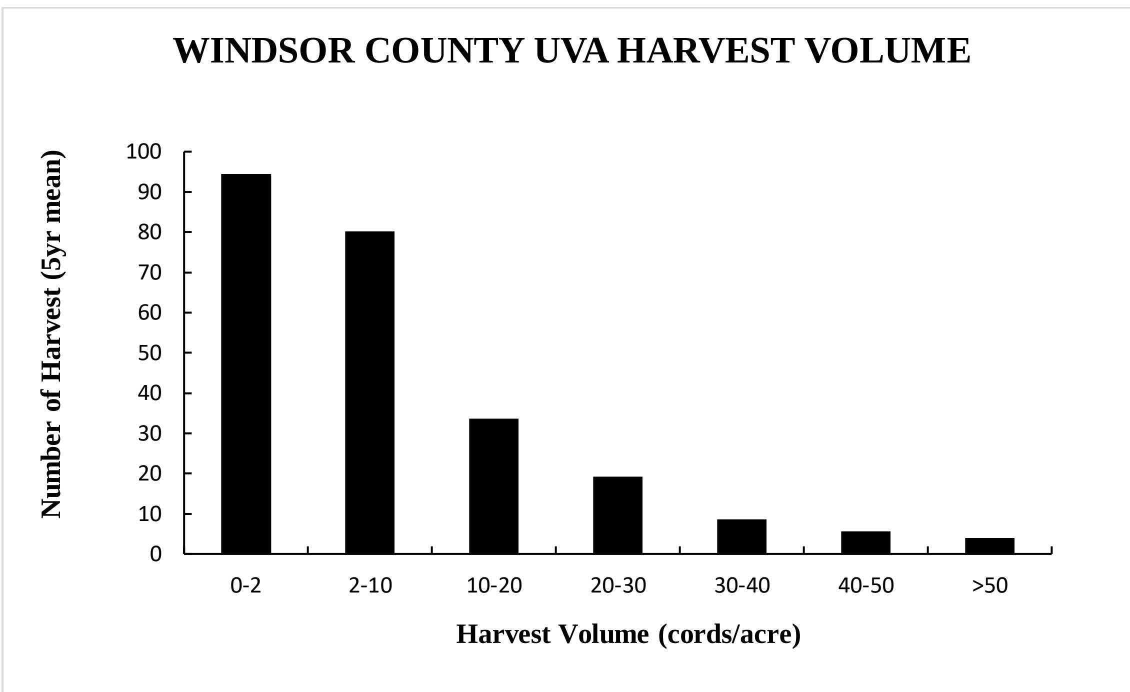


Figure 2: Distribution of harvest volumes (cords/acre) for Windsor county based on a five year mean 2012-2016.

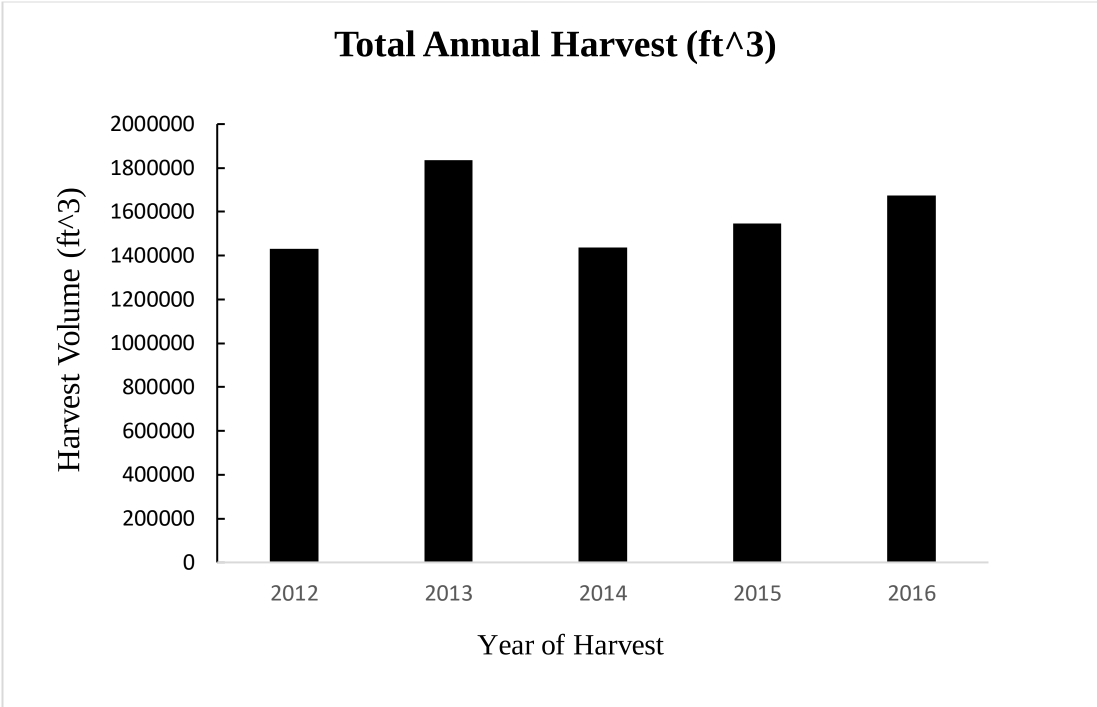


Figure 3: Total annual harvest volumes (ft^3) reported by FMARs in Windsor county for a five-year period.

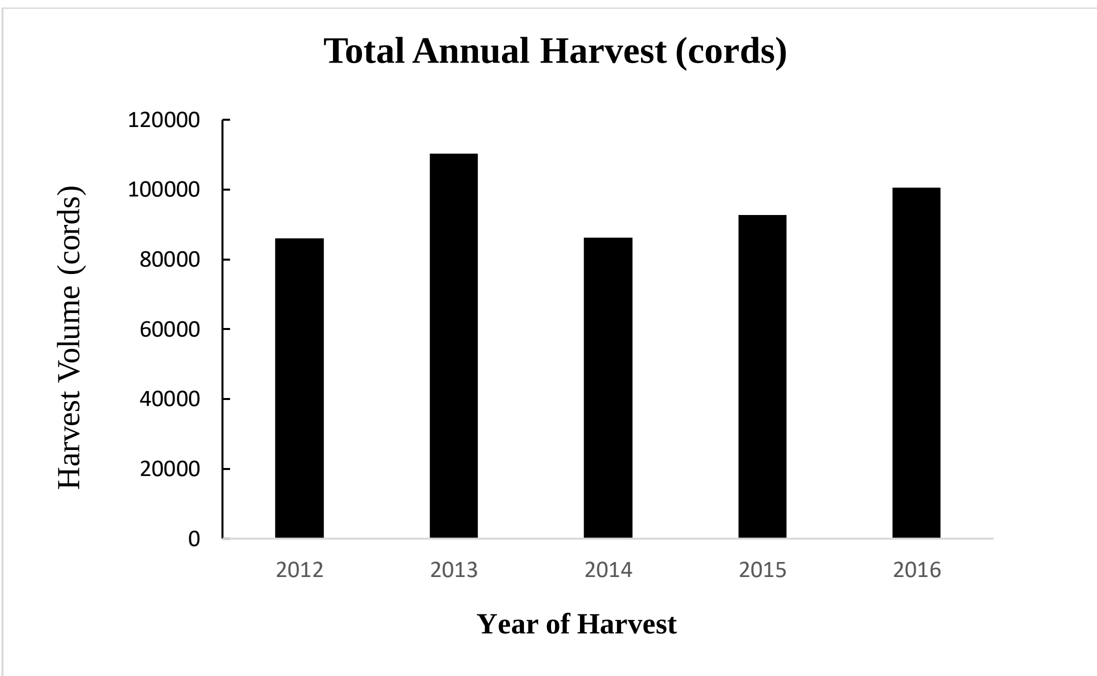


Figure 4: Total annual harvest volumes (cords) reported by FMARs in Windsor county for a five-year period.

Discussion

These results help inform modeling efforts aimed at investigating how current forest management may impact future forest conditions (structure, composition, function). These findings highlight the expected harvest intensities for the region and further inform the selection of management approaches for the landscape. In another similar investigation of UVA data, we looked at the frequency of prescribed management types across all enrolled UVA parcels in Windsor County and across different forest types (*Windsor County VT, UVA Stand Data: Management Type Summary October 18, 2017 – Matthias Nevins).* We found that single tree selection was the most commonly prescribed silvicultural treatment followed by intermediate thinning. These finding are supported by this investigation of reported harvest volumes. With average harvest rates of 125.8 ft^3/acre or 7.6 cords/acre we see that the average harvest rates are low. This would appear to coincide with silvicultural treatments which typically retain more standing volume. Table 2 below outlines typical silvicultural treatments as they relate to the harvest rates reported here.

The harvest rates reported here are based on the entire parcel area and not the stand area associated with the harvest. Therefore, the volumes are lower than is expected for a typical harvest. However, we can use these results to make assumptions about the types of management that is occurring. Low volume coincides with partial cutting and thinning while higher volumes are associated with treatments such as clearcutting.

Table 2: Harvest rates and associated silvicultural methods

Harvest Volume (ft^3/acre)	**Cords/Acre**	**Silvicultural method(s)**
0-25	0-2	Thinning & Single tree selection
25-100	2-10	Single tree and Group selection
100-200	10-20	Group selection, Shelterwood
200-300	20-30	Large Group selection, Shelterwood, Seed tree
300-400	30-40	Overstory removal, Clearcutting
>500	>50	Clearcutting